Appendix C

Water Quality Supporting Technical Information

C.1   Water Temperature

C.1.1   Upper Klamath Basin

C.1.1.1 Keno Impoundment/Lake Ewauna

Water temperatures in much of the Upper Klamath River, including the reach from Link River Dam through the Keno Impoundment/Lake Ewauna, exceed 20°C (68°F) in June through August. The Keno Impoundment/Lake Ewauna experiences periods of intermittent, weak summertime stratification, however water temperatures are generally similar throughout the water column and are among the warmest in the Klamath Basin with peak values greater than 25°C (77°F). Weekly measurements in 2007 in the Link River and upper portion of the Keno Impoundment indicate maximum temperatures of 25ºC (77ºF) in mid-to-late summer (Deas and Vaughn 2006, Sullivan et al. 2008). Recorded average monthly temperatures for the period 2001 to 2004 in Keno Impoundment were 22.4ºC (72.3ºF) in July, 20.8ºC (69.4ºF) in August and 18.0ºC (64.4ºF) in September (FERC 2007). Average monthly temperatures reported by PacifiCorp downstream from Keno Dam for the same period were 23.2ºC (73.8ºF), 21.1ºC (70.0ºF), and 16.9ºC (62.4ºF) during July, August, and September, respectively (FERC 2007). Similarly, during 2009, summer water temperatures downstream from Keno Dam were generally greater than 16 °C (60.8 °F) from June through September, with peak temperatures exceeding 26 °C (78.8 °F) in late-July (Watercourse Engineering, Inc. 2011a).

C.1.1.2 Hydroelectric Reach

The Hydroelectric Reach spans the Oregon–California state line from J.C. Boyle to Iron Gate Dam. During summer months, maximum weekly maximum temperatures (MWMTs) in the Hydroelectric Reach regularly exceed the range of chronic effects temperature thresholds (13 to 20ºC [55.4 to 68ºF]) for full salmonid support (North Coast Regional Board 2010, Kirk et al. 2010, Asarian and Kann 2011).

In general, water temperatures in this reach follow a seasonal pattern, with average monthly water temperatures from March through November ranging from just over 5°C (41ºF) in November to more than 22°C (71.6°F) during June through August (FERC 2007). Winter water temperatures throughout the reach are largely driven by the temperature of river inflows (Deas and Orlob 1999). In the summer, the relatively shallow J.C. Boyle Reservoir does not exhibit long-term thermal stratification, with a typical vertical temperature difference of less than 2°C (3.6ºF) in the water column (FERC 2007; Raymond 2008, 2009, 2010). In the Bypass Reach downstream from J.C. Boyle Dam, at approximately RM 225.8, water from cold groundwater springs enters the river at a relatively constant 11 to 12ºC (51.8 to 53.6ºF). During daily peaking operations at J.C. Boyle Powerhouse (RM 225.2), warmer reservoir discharges are diverted around this reach (see also Section 2.3.1 *J.C. Boyle Dam Development*), leaving the cold groundwater springs to dominate the river flow. Water temperatures in the Bypass Reach can decrease by 9–27°F when bypass operations are underway due to the influence of the springs (Kirk et al. 2010). Downstream in the Peaking Reach, the cooler, spring-influenced river mixes rapidly with the warmer water discharged from the J.C. Boyle Powerhouse that can exceed 25ºC (77ºF) in July and August (Kirk et al. 2010). In the Peaking Reach, downstream of the Bypass Reach, the flow diverted around the Bypass Reach rejoins the Klamath River (see Figure 2.3-1). At the upstream

end of the Peaking Reach, the natural, cold groundwater input into the Bypass Reach, combined with fluctuations in river flow due to hydroelectric power operations in the Peaking Reach also produces an observed increase in daily water temperature range above the natural diel water temperature fluctuations (Kirk et al. 2010).

Further downstream in the Peaking Reach, near the confluence of the Klamath River and Shovel Creek (Figure 2.2-3), there are natural hot springs that contribute flows to the mainstem river. The natural hot springs were not found to result in warming of the Klamath River based on two measurements made in November and December 2017. Water temperature data collected upstream and downstream of the confluence of the Klamath River and Shovel Creek showed a 1.4°F increase in the downstream direction during the November 2017 measurement, but a 0.2°F decrease during the December 2017 measurement (KRRC 2018).

Within and downstream from Copco No. 1 Reservoir, spring, summer and fall temperatures in the Hydroelectric Reach are heavily influenced by the large thermal mass of the two deepest reservoirs, Copco No. 1 and Iron Gate reservoirs, and their seasonal stratification patterns. Spring temperatures are generally cooler than would be expected under natural conditions, and summer and fall temperatures are generally warmer (PacifiCorp 2004a, North Coast Regional Board 2010). Both Iron Gate and Copco No. 1 reservoirs thermally stratify beginning in April/May and do not mix again until October to December (Figure C-1, Table C-1) (Raymond 2008, 2009, 2010; Asarian and Kann 2011). Water temperature data indicate thermal stratification in Copco No. 1 Reservoir begins about a month later and ends about a month earlier than stratification in Iron Gate Reservoir (Asarian and Kann 2011). The onset of spring/summer stratification and the timing of fall turnover in Iron Gate and Copco No. 1 Reservoirs are driven by meteorological conditions (Deas and Orlob 1999, Asarian and Kann 2011).
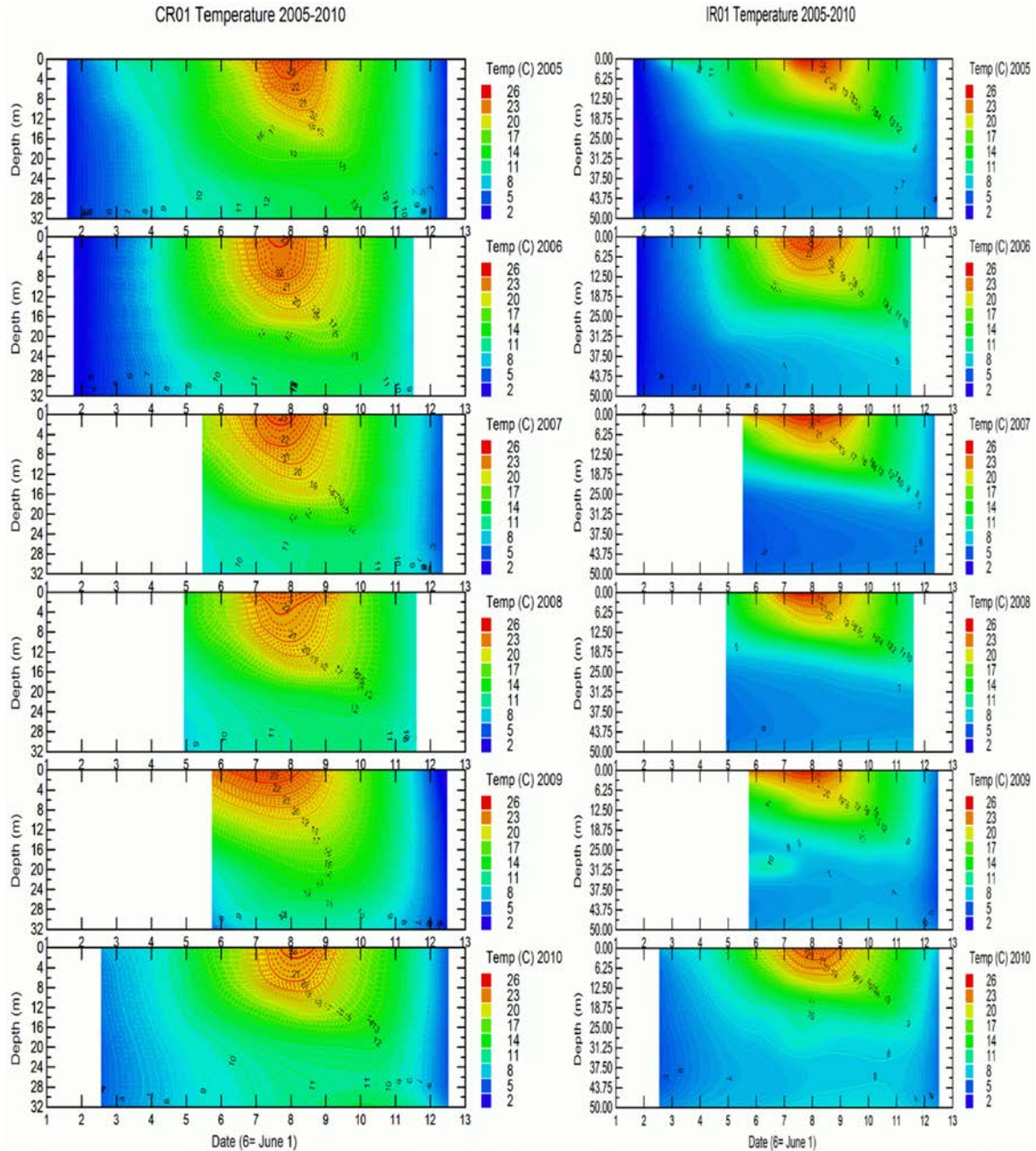
Figure C-1. Depth-time Distribution of Isopleths of Water Temperature at Station CR01 in Copco No. 1 Reservoir and IR01 in Iron Gate Reservoir from January 2005 to December 2010. Source: Asarian and Kann 2011.

Table C-1. General Reservoir Turnover Dates for Copco No. 1 and Iron Gate Reservoirs (2007 to 2009).

Year	Thermally Stable Hypolimnion Establishment Date		Approximate Reservoir Turnover Date		Source
	Copco	**Iron Gate**	**Copco**	**Iron Gate**	
2007	By June 6	By June 6	Before October 23	Before November 28	Raymond 2008
2008	By April 30	By April 30	Before October 22	Before November 19	Raymond 2009
2009	By May 24	By May 24	Before October 13	Before November 17	Raymond 2010

Powerhouse withdrawals for Copco No. 1 and Iron Gate dams are primarily from the epilimnion (surface water). The depth of the epilimnion changes over time based on the season and characteristics of each reservoir. The depth of the thermocline (metalimnion) separating the epilimnion (surface waters) from the hypolimnion (bottom waters) is approximately 50 feet below the water surface in both Copco No. 1 and Iron Gate reservoirs by mid-summer when thermal stratification is present in the reservoirs (FERC 2007). In Copco No. 1 Reservoir, powerhouse withdrawal is from approximately 9.8 m (32 ft) below the water surface when the reservoir is full (full pool) and in Iron Gate Reservoir powerhouse withdrawal is from approximately 10.7 m (35 ft) below the water surface when the reservoir is full (FERC 2007). Occasionally, withdrawals extend into the hypolimnion; for example, in Iron Gate Reservoir, the withdrawal envelope has been estimated to extend down to approximately 18 m (60 ft) in depth (Deas and Orlob 1999). Additionally, a small withdrawal (about 50 cfs) for the Iron Gate Hatchery occurs from the hypolimnion at Iron Gate Reservoir. In general, however, temperature in waters discharged from Copco No. 1 and Iron Gate reservoirs reflect the warmer temperatures of surface water (NRC 2003). Seasonal stratification of these two reservoirs also prevents mixing of waters within the water column and adversely affects dissolved oxygen concentrations, nutrient concentration (and speciation), and pH in bottom waters, limiting the potential for hypolimnetic cool water releases to the Mid- and Lower Klamath River (FERC 2007). The small relative volumes of the hypolimnions in Copco No. 1 and Iron Gate reservoirs also limit the potential for seasonal releases to decrease water temperatures in downstream river reaches. Since J.C. Boyle Reservoir does not exhibit long-term thermal stratification (i.e., it lacks a seasonal hypolimnion), there are no controllable actions that can be taken to cool water released from this waterbody (FERC 2007).

C.1.2   Mid- and Lower Klamath Basin

C.1.2.1   Iron Gate Dam to Salmon River

Water temperature in the Lower Klamath Basin varies seasonally, with mean monthly temperatures in the river downstream from Iron Gate Dam ranging from 3 to 6°C (37 to 43°F) in January to 20 to 22.5°C (68 to 72.5°F) in July and August (Bartholow 2005; Karuk Tribe of California 2009, 2010a, 2010b, 2011, 2012, 2013; Watercourse Engineering, Inc. 2011a, 2011b, 2012, 2013, 2014, 2015, 2016; Asarian and Kann 2013). Water temperature increases with distance downstream from Iron Gate Dam due

to meteorological controls (Basdekas and Deas 2007, Asarian and Kann 2013). Based upon annual water temperature monitoring conducted by the Karuk Tribe, water temperatures peak during the summer when air temperatures increase and flows decrease in the Klamath Basin (Figure C-2; Karuk Tribe of California 2002, 2003, 2007, 2009, 2010a, 2010b, 2011, 2012, 2013). Figures C-2, C-3, C-4, C-5, and C-6 show the range of annual variability in daily average water temperature at several Klamath River locations downstream of Iron Gate Dam from 2006 to 2013. Daily average summer water temperatures regularly exceed 20°C during summer months near Seiad Valley, while daily average values documented immediately downstream from Iron Gate Dam are generally cooler and less variable. Daily average temperatures between June and September are between 1 to 6°C (1.8 to 10.8°F) higher near Seiad Valley than those just downstream from Iron Gate Dam (Figure C-2 to C-6; Karuk Tribe of California 2009, 2010a, 2010b, 2011, 2012, 2013). Water temperature modeling in the Klamath River indicates releases from Iron Gate Dam do not affect mainstem water temperature by the Salmon River (RM 66.3) (PacifiCorp 2005, Dunsmoor and Huntington 2006, North Coast Regional Board 2010, Perry et al. 2011, Risley et al. 2012).

Water temperature trends under the 2013 BiOp flows are consistent with measured water temperature between 2001 and 2012 (Watercourse Engineering, Inc. 2011a, 2011b, 2012, 2013, 2014, 2015 2016; Asarian and Kann 2013). An average of water temperature data from 2001 to 2011 at locations along the Klamath River shows that daily mean and daily maximum water temperature peaked between July and August with a maximum temperature of approximately 24°C (Figure C-7; Asarian and Kann 2013). A comparison of water temperature and flow measured in 2009 (Figure C-8) and 2015 (Figure C-9) demonstrates that water temperature downstream of Iron Gate Dam in 2015 peaked in July under 2013 BiOp minimum flows of 900 cfs (NMFS and USFWS 2013, Watercourse Engineering, Inc. 2016), similar to the water temperature downstream of Iron Gate Dam in 2009 which peaked between July and August under 2002 Biological Opinion minimum flows of 1,000 cfs (NMFS 2002, Watercourse Engineering, Inc. 2010). Similar water temperature trends are also observed in the Klamath River at Seiad Valley (Figures C-8 and C-9).

With respect to the longer term water temperature record (i.e., prior to 2000), Bartholow (2005) presents evidence that water temperatures in the lower Klamath River have been increasing since before 1950. Bartholow (2005) indicates that the observed multi-decade trend of increasing water temperatures in the lower river is related to the cyclic Pacific Decadal Oscillation and is consistent with a measured average basin wide air temperature increase of 0.33°C/decade (0.59°F/decade). Bartholow (2005) estimates that the season of high temperatures that are potentially stressful to salmonids has lengthened by about 1 month in the Klamath River since the early 1960s, and the average length of the lower river exhibiting summer water temperatures less than 15°C (59°F) has declined by about 8.2 km/decade (5.1 mi/decade). Potential climate change effects on water temperature are discussed in more detail as part of the effects determination for the No Project Alternative (see Section 3.2.4.3).
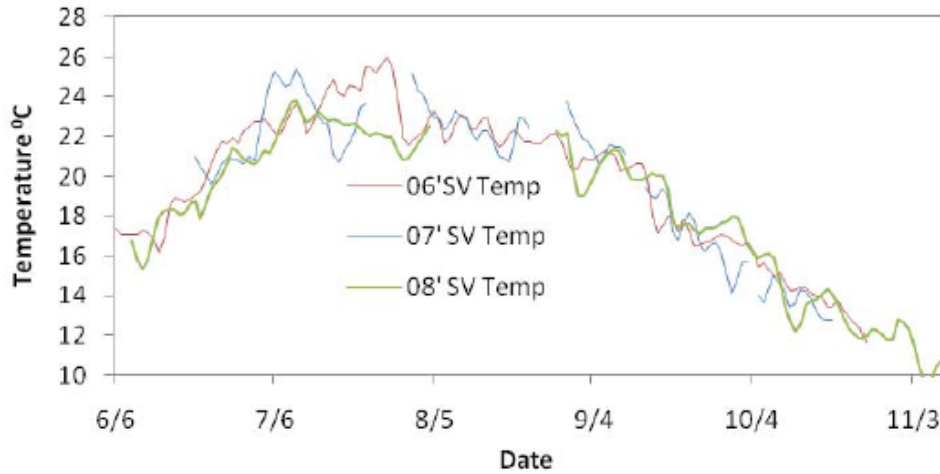
Figure C-2. Daily Average Water Temperature in the Klamath River near Seiad Valley (RM 132.7) June through November 2006, 2007, and 2008. Source: Karuk Tribe of California 2009.
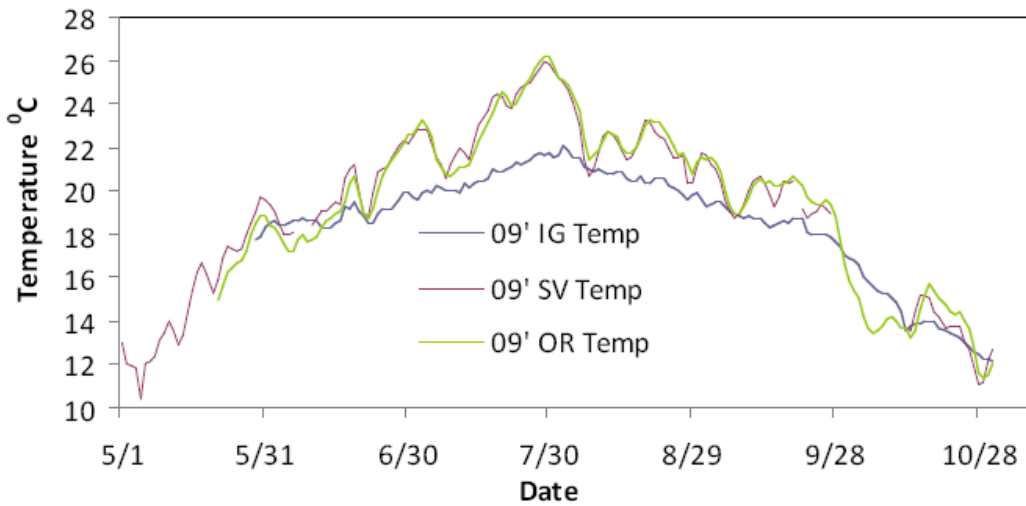

Figure C-3. Daily Average Water Temperature in the Klamath River Downstream from Iron Gate Dam (≈RM 193.1), near Seiad Valley (RM 132.7) and at Orleans (RM 58.9) During May through October 2009. Source: Karuk Tribe of California 2010a.
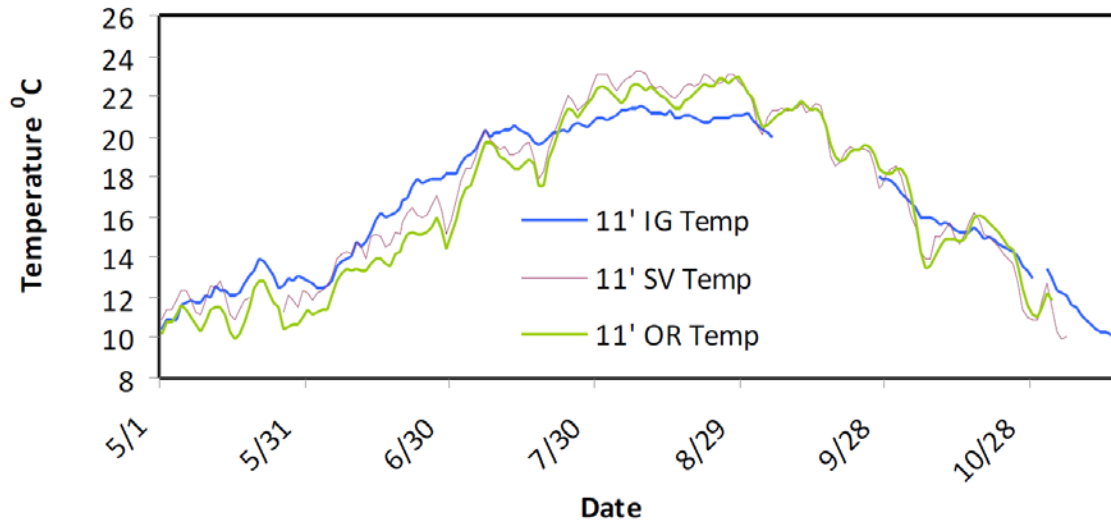
Figure C-4. Daily Average Water Temperature in the Klamath River Downstream from Iron
Gate Dam (≈RM 193.1), near Seiad Valley (RM 132.7) and at Orleans (RM 58.9)
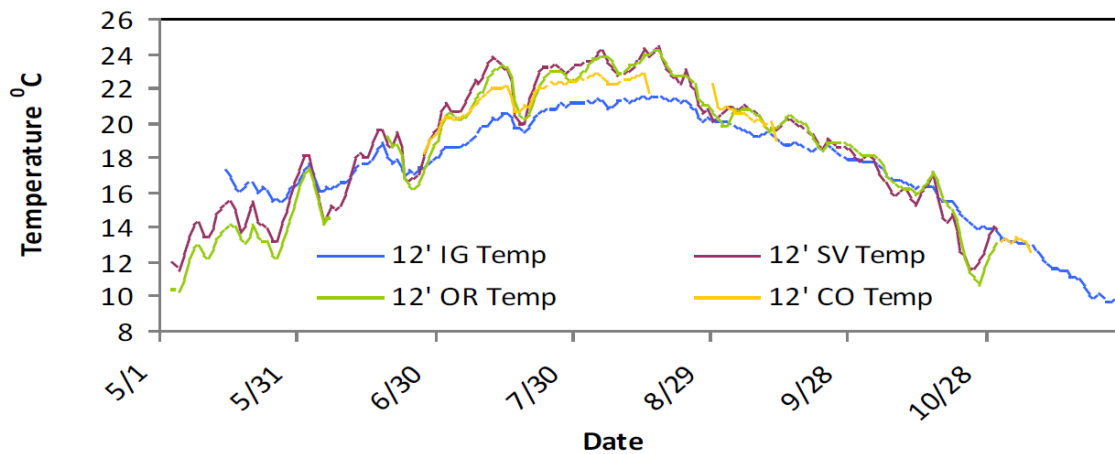During May through November 2011. Source: Karuk Tribe of California 2011.

Figure C-5. Daily Average Water Temperature in the Klamath River Downstream from Iron
Gate Dam (≈RM 193.1), near Colliers Rest Area/I-5 Bridge upstream of the Shasta
River confluence (≈RM 179.5), near Seiad Valley (RM 132.7) and at Orleans (RM
58.9) During May through November 2012. Source: Karuk Tribe of California 2012.
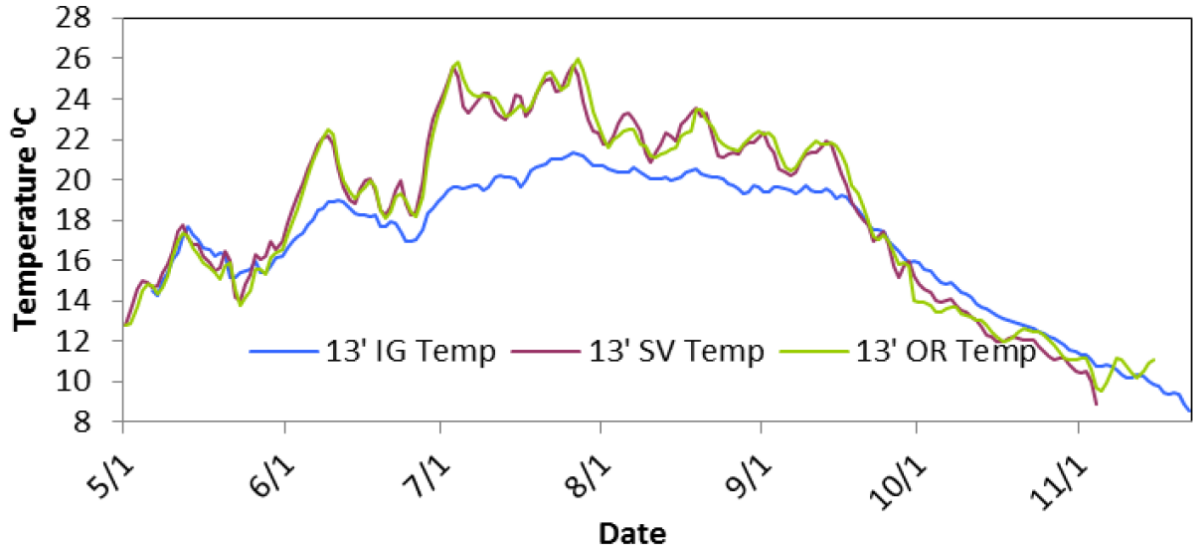
Figure C-6. Daily Average Water Temperature in the Klamath River Downstream from Iron
Gate Dam (≈RM 193.1), near Seiad Valley (RM 132.7) and at Orleans (RM 58.9)
During May through November 2013. Source: Karuk Tribe of California 2013.
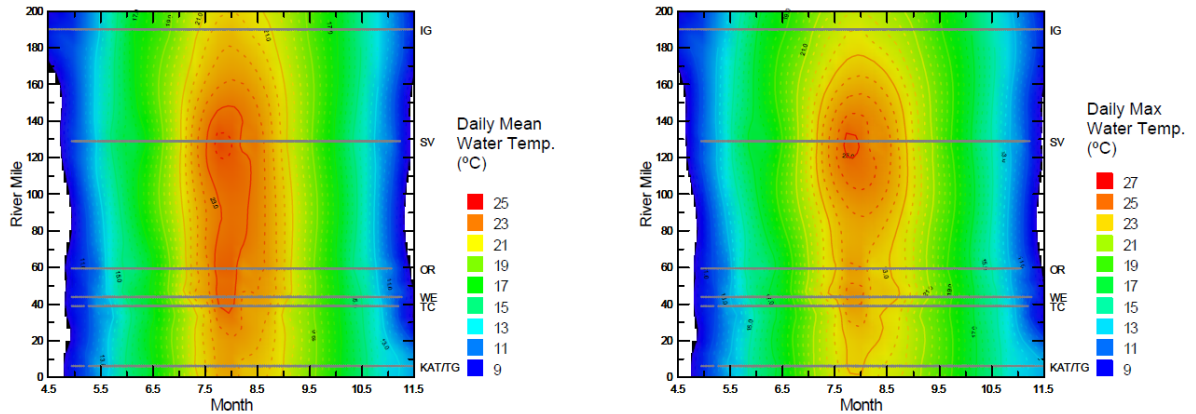
Figure C-7. Daily Mean and Daily Maximum Water Temperature in the Klamath River
Downstream from Iron Gate Dam (≈RM 193.1) to the Klamath River at Turwar (≈RM
5.6) Averaged from 2001 to 2011 Data. Horizontal Grey Lines Indicate the
Location and Time Period of Measurement in the Klamath River at Iron Gate (IG),
Seiad Valley (SV), Orleans (OR), Weitchpec Upstream of the Trinity River (WE),
Upstream of Tully Creek (TC), and Turwar (KAT/TG). Source: Asarian and Kann
2013.

Figure C-8. Klamath River Water Temperature in 2009 During Monthly Water Quality Grab Samples and Flow at USGS Gage Stations (Station Number Listed in the Legend). Source: Watercourse Engineering, Inc. 2011a.

Figure C-9. Klamath River Water Temperature in 2015 During Monthly Water Quality Grab Samples and Flow at USGS Gage Stations. Source: Watercourse Engineering, Inc. 2016.

C.1.2.2   Salmon River to Estuary

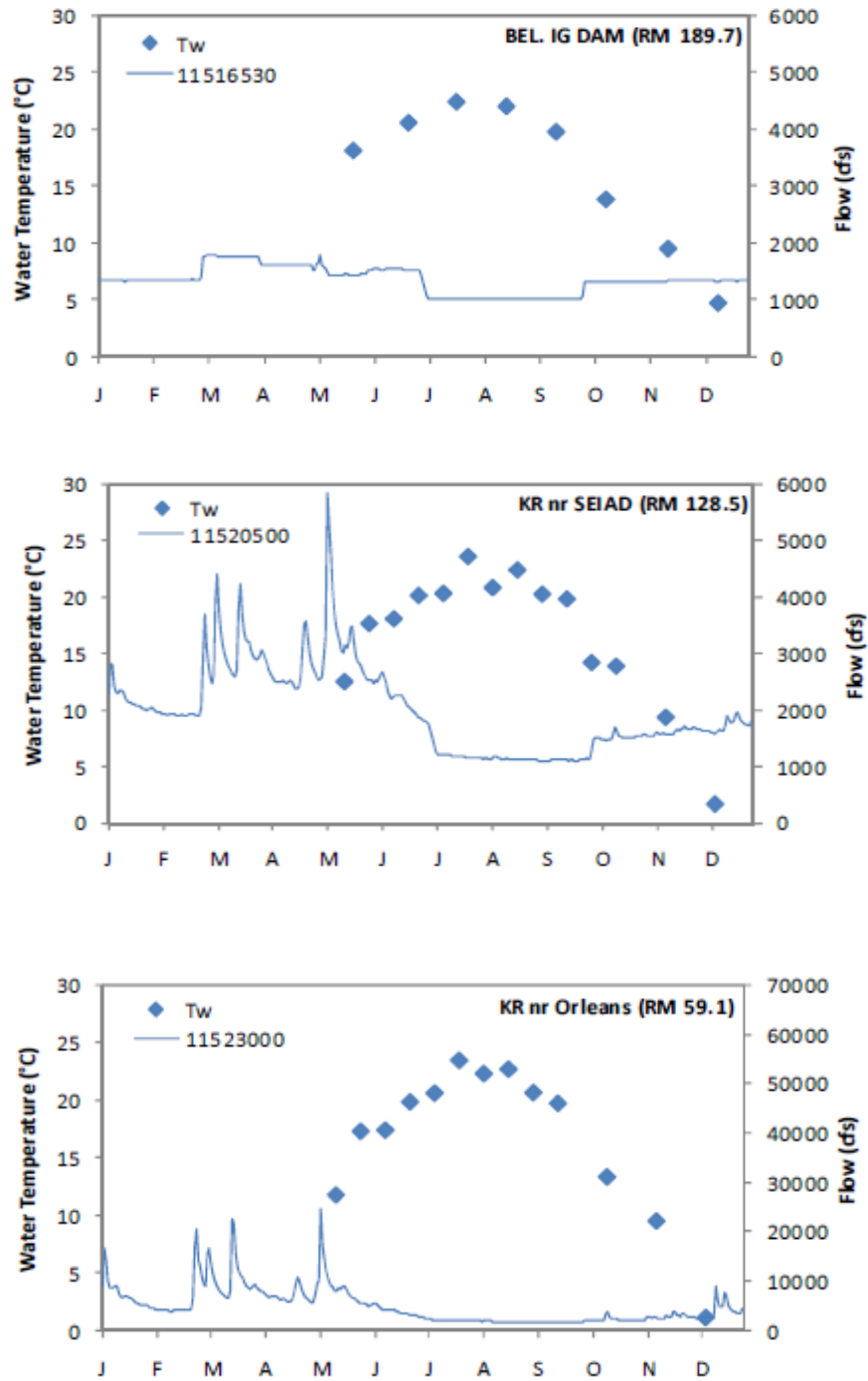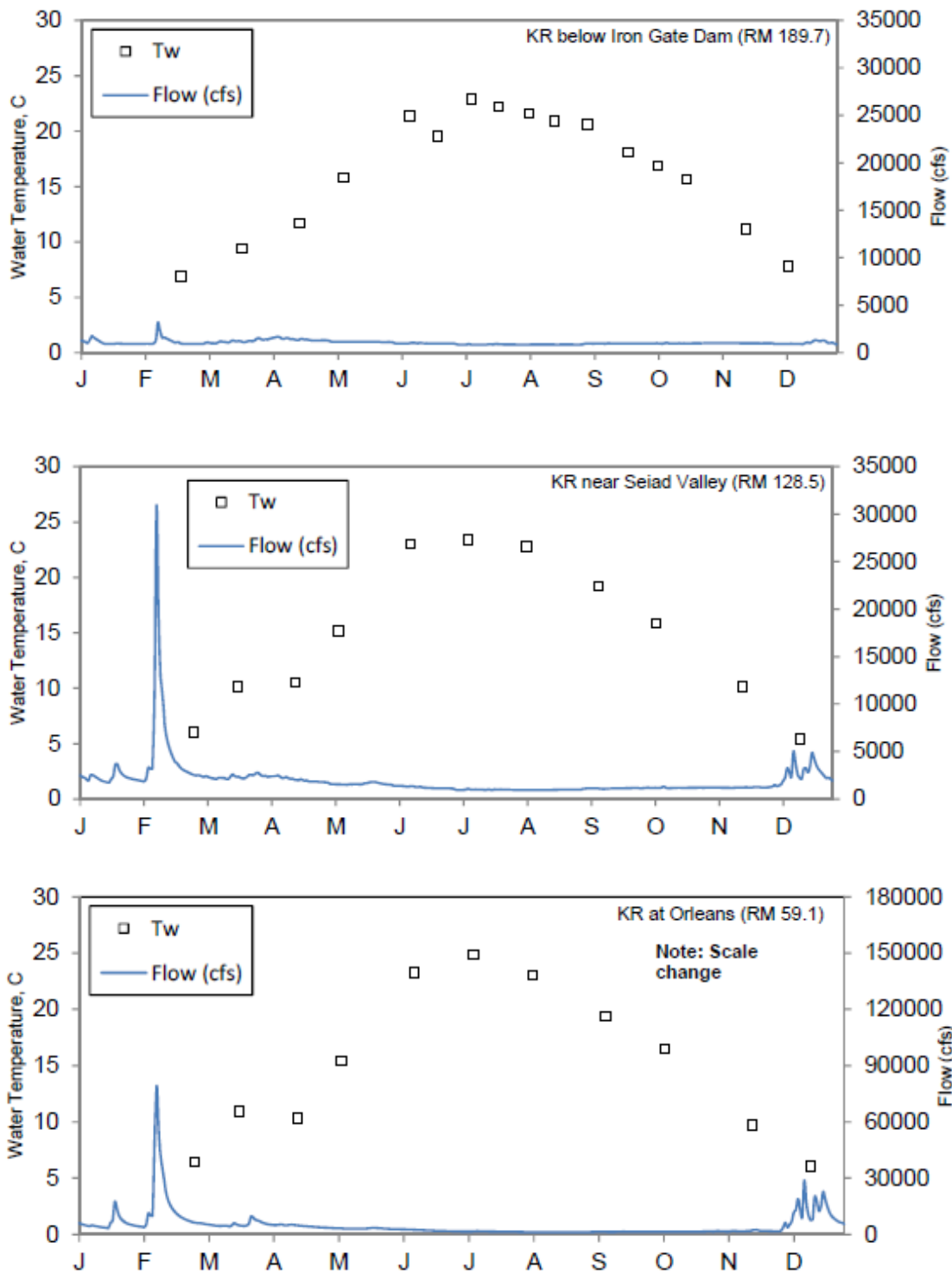Water temperature monitoring by the Karuk Tribe includes data from Orleans (RM 58.9), which is just downstream from the Salmon River confluence with the mainstem Klamath River. Daily average water temperature at Orleans was 10.5 to 26°C (50.9 to 78.8°F) from June through November 2006 to 2008, with the warmest temperatures generally occurring during July (Figure C-10; Karuk Tribe of California 2009). More contemporary data from 2009 to 2015 (Figures C-3 to C-9) further support these water temperature trends at Orleans (Karuk Tribe of California 2009, 2010a, 2010b, 2011, 2012, 2013; Watercourse Engineering, Inc. 2011a, 2011b, 2012, 2013, 2014, 2015, 2016; Asarian and Kann 2013). Asarian and Kann (2013) reports that average daily maximum water temperature between 2001 and 2011 is approximately 23°C to 24°C between July and August from the Salmon River (RM 66.3) to Turwar Creek (RM 5.6) (Figure C-7).

In the mainstem river between the Klamath River's confluence with the Trinity River and the Klamath River Estuary, the Yurok Tribe, through the Yurok Tribe Environmental Program (YTEP), has conducted annual water temperature monitoring since 2002 (YTEP 2004; Sinnott 2010a, 2011a, 2012a; Hanington 2013; Hanington and Ellien 2013). Between 2009 and 2013, peak temperatures generally occur in mid-July to mid-August with the highest daily maximum temperatures recorded at the most upstream locations (Sinnott 2010a, 2011a, 2012a; Hanington 2013; Hanington and Ellien 2013). The effect of the Trinity River on the mainstem Klamath River water temperature varied within individual years and between years. A small (0.5°C [0.9°F] or less) cooling effect was usually observed from the contribution of the Trinity River to the mainstem Klamath River between 2009 and 2011, but inflows from the Trinity River caused a 0.5°C (0.9°F) or less warming effect between mid-May and late June during 2010 and 2011 (Sinnott 2010a, 2011a, 2012a). During 2012 and 2013, inflows from the Trinity River usually altered water temperature in the Klamath River by 0.5°C (0.9°F) or less, but from mid-August to late September inflow from the Trinity River cooled Klamath River water temperature on average by approximately 1°C (Hanington 2013, Hanington and Ellien 2013). During May through November 2009, water temperatures ranged from approximately 11.1°C (52.0°F) in October to 26.8°C (80.2°F) in July (Sinnott 2010a). Similar trends were measured during 2013 with the lowest water temperature of 12.4°C (54.2°F) occurring in October while the highest water temperature of 26.4°C (79.4°F) occurred in July (Hanington and Ellien 2013). Between 2009 and 2013, the daily maximum summer water temperatures ranged from approximately 23.8 to 26.9°C (74.8 to 80.4°F) just upstream of the confluence with the Trinity River (Weitchpec [RM 43.3]), decreasing to approximately 23.4 to 24.9°C (74.1 to 76.8°F) near Turwar Creek (RM 5.6) (YTEP 2005; Sinnott 2010a, 2011a, 2012a; Hanington 2013; Hanington and Ellien 2013). Figure C-11 shows the upper range of the daily maximum water temperature which occurred in 2009, while Figure C-12 shows the lower range of the daily maximum water temperature which occurred in 2011 (Sinnott 2010a, 2012a). These summer temperatures exceed optimal growth thresholds as well as critical thermal maxima for coho, Chinook salmon, and steelhead (Brett 1952, Armour 1991, Stein et al. 1972, McGeer et al. 1991). Historically, summer water temperature maxima in the lower Klamath River have been greater than in other coastal rivers to the north and south. For example, Blakey (1966, as cited in Bartholow 2005), reports water temperatures in the Klamath River downstream from the Trinity River confluence (RM 43.3) reaching 26.6°C (79.9°F) for up to 10 days per year, in contrast to proximal coastal rivers that never reach this temperature.
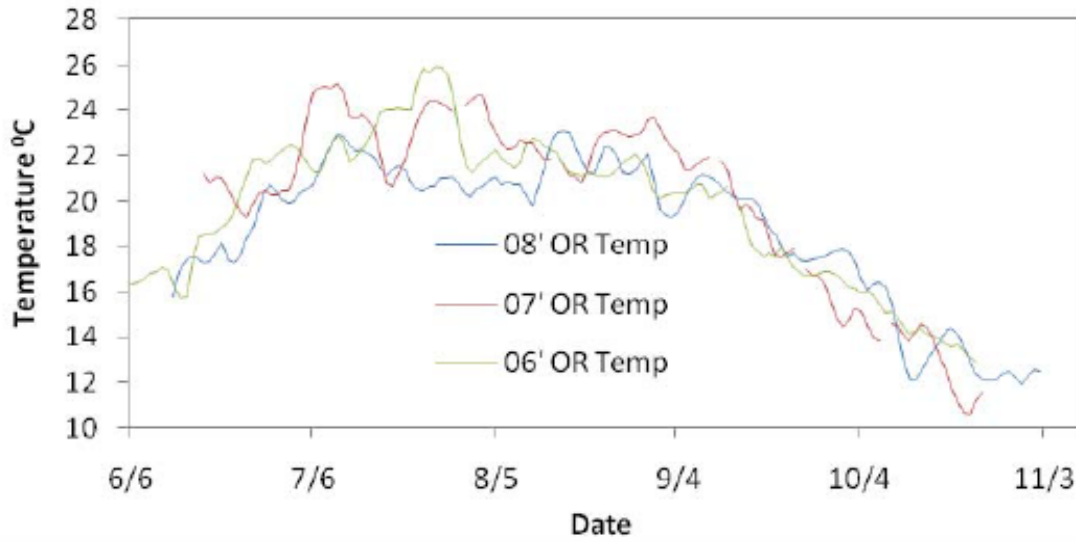
Figure C-10. Daily Average Water Temperature in the Klamath River at Orleans (RM 58.9) June through November 2006, 2007, and 2008. Source: Karuk Tribe of California 2009.
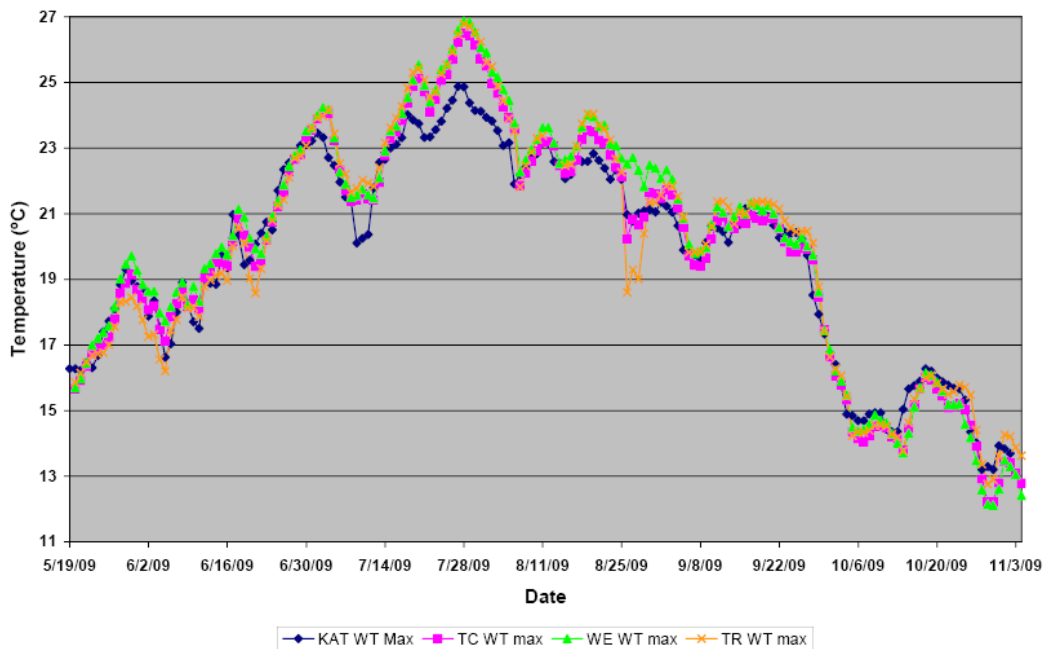
Figure C-11. Daily Maximum Water Temperatures in the Klamath River at Weitchpec (RM 43.6 [WE]), Upstream of Tully Creek (RM 40.1 [TC]), and Upstream of Turwar Boat Ramp (RM 6 [KAT]), as well as in the Trinity River (RM 43.3) near the Confluence with the Klamath River (RM 0.5 [TR]) May through November 2009. Source: Sinnott 2010a.
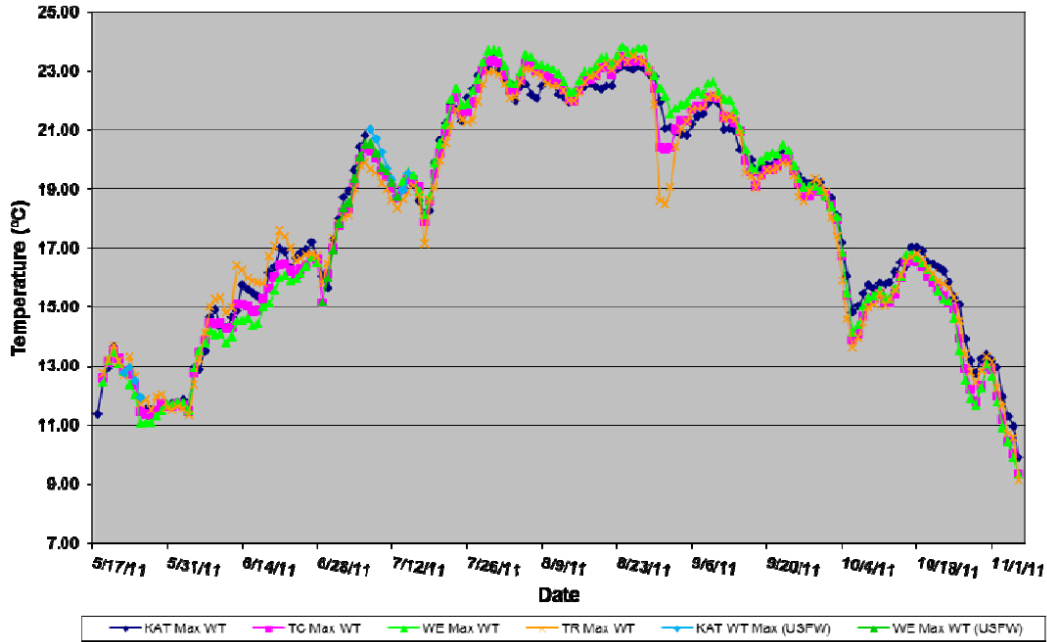
Figure C-12. Daily Maximum Water Temperatures in the Klamath River at Weitchpec (RM 43.6
[WE]), Upstream of Tully Creek (RM 40.1 [TC]), and Upstream of Turwar Boat
Ramp (RM 6 [KAT]), well as in the Trinity River (RM 43.3) near the Confluence
with the Klamath River (RM 0.5 [TR]), at Upstream of Turwar Boat Ramp from the
U. S. Fish and Wildlife (RM 6 [KAT (USFW)]), and at Weitchpec from U. S. Fish
and Wildlife (RM 43.6 [WE (USFW)]) May through November 2011. Source: Sinnott
2012a.

C.1.2.3 Klamath River Estuary

Hydrodynamics and water quality within the Klamath River Estuary are highly variable
spatially and temporally and are greatly influenced by season, river flow, vertical water
column stratification (thermal and/or chemical), and location of the estuary mouth, the
latter changing due to periodic sand bar movement. Input of cool ocean water and fog
along the coast minimizes extreme water temperatures much of the time (Scheiff and
Zedonis 2011). Water temperature has been monitored in the Klamath River Estuary by
California Department of Fish and Game (Wallace 1998) and most recently by the Yurok
Tribe Fisheries Program (Hiner 2006) and the YTEP (2005), with support from the North
Coast Regional Board. Water temperatures in the Klamath River Estuary from
December through April are roughly 5 to 12°C (41 to 54°F) (Hiner 2006). In summer and
fall months, warmer air temperatures and lower flows result in increased water
temperatures. Under low-flow summertime conditions, water temperatures in the
Klamath River Estuary have been observed at 20 to 24°C (68 to 75.2°F) (Wallace 1998)
or greater than 24°C (75.2°F) (Hiner 2006). During June to September from 2009 to
2015, water temperatures during water quality grab samples ranged from approximately
13.1 to 21.9°C (55.6 to 71.4°F) (Watercourse Engineering, Inc. 2011a, 2011b, 2012,
2013, 2014, 2015, 2016). These levels exceed optimal growth thresholds for salmonids,
as cited in the previous section.

Estuarine water temperature is linked to upstream hydrology and periods of mouth closure because when the estuary mouth is open, denser salt water from the ocean sinks below the less dense fresh river water, resulting in chemical stratification and a "salt wedge" that moves up and down the estuary with the daily tides (Horne and Goldman 1994, Wallace 1998, Hiner 2006). The salt wedge is also thermally stratified with cooler, higher salinity ocean waters remaining near the estuary bottom, and warmer, lower salinity river water near the surface. Upstream hydrology can affect the location of the salt water wedge and thus, affect thermal structure in the Klamath River Estuary. For example, during pulse flows released from the Lewiston Dam on the Trinity River in August 2004, the upstream extent of the salt wedge moved downstream approximately one mile (YTEP 2005). In the Klamath River Estuary, mouth closure has been reported to reduce the size of the salt water wedge, decrease overall salinity, and subsequently increase water temperatures in the Klamath River Estuary (Hiner 2006). Mouth closure, caused by formation of a sand berm across the mouth of the Klamath River Estuary, is a function of off-shore and alongshore wave power and sediment supply, freshwater inflows, the tidal prism, and morphological characteristics of the inlet (Escoffier 1940, Brunn 1966, O'Brien 1971, Barnes 1980). The historical frequency and duration of mouth closure in the Klamath River Estuary has not been documented, although it is expected to occur during low-flow periods (June to October).

C.2   Suspended Sediments

For the purposes of the Lower Klamath Project EIR, "suspended sediments" refer to settleable suspended material in the water column. Bed materials, such as gravels and larger substrates, are discussed in Section 3.11.2.4 *Sediment Load*. Two types of suspended sediments are considered for the analysis in the Lower Klamath Project EIR: algal-derived (organic) suspended material and mineral (inorganic) suspended material. Sources of each type of suspended material differ, as do spatial and temporal trends for each within the Upper, Mid-, and Lower Klamath Basins.

Suspended sediments in the water column are quantified using different methods, depending on the study. Two standard methods widely used for determining the amount of suspended sediments in water are Method D 3977-97, which reports suspended sediments as suspended sediment concentrations (SSCs), and Method 2450 D, which reports the suspended sediments as total suspended solids (TSS) (Gray et al. 2000). While data produced by the SSC method are more representative of natural suspended sediments, SCC is considered equivalent to TSS for this report. As needed, data from multiple sources, reported as either TSS or SSC, are used interchangeably for the Lower Klamath Project EIR, despite potential differences in the numeric values reported by each method (Gray et al. 2000).

Turbidity, an optical property referring to the amount of light scattered or absorbed by a fluid, is another common way to quantify suspended sediments and is measured in nephelometric turbidity units (NTUs). The exact relationship between turbidity and suspended sediment is dependent on the parent geology and must be determined for each watershed (Montgomery 1985; MacDonald et al. 1991). High suspended sediments in the water column affect organisms directly (e.g., interfering with vision) or indirectly by changing water temperature (e.g., suspended sediment particles absorb heat from sunlight) and reducing dissolved oxygen (DO) concentrations by scattering light and reducing photosynthetic activity. Suspended sediments are also a water quality

concern because they are often associated with storing contaminants from the water column (e.g., polar organics and cationic metal forms). Municipal and domestic water supply beneficial uses can also be adversely affected by changes in suspended sediment concentrations and turbidity in streams.

For the Klamath River, coincident turbidity data is occasionally presented along with TSS data. However, as the dataset is not consistent in space or time, turbidity levels are not used to support significance determinations (see Section 3.2.4 *Impact Analysis Approach*) and are not analyzed in detail in the Lower Klamath Project EIR.

C.2.1 Upper Klamath Basin

C.2.1.1 Hydroelectric Reach

Suspended sediment generally decreases through the Hydroelectric Reach as suspended sediment from upstream is intercepted, decomposed, retained, or diluted between the J.C. Boyle and Copco No. 1 reservoirs. Organic suspended sediment originating from Upper Klamath Lake is the predominant form of suspended sediment entering the Hydroelectric Reach from upstream and affecting water quality (PacifiCorp 2004a, Deas and Vaughn 2006, Watercourse Engineering, Inc. 2011). During the winter and spring (November through April), the reservoirs at the Lower Klamath Project intercept and retain inorganic suspended sediment delivered from tributaries (e.g., Shovel Creek, Fall Creek, Jenny Creek) to the reservoirs, where peak concentrations occur in association with high-flow events. While this may be somewhat beneficial for downstream reaches by decreasing suspended sediment concentrations and turbidity, the interception of inorganic sediments by the reservoirs does not appear to be an important mechanism related to sediment delivery in the mainstem Klamath River. This is because a relatively small (3.4 percent) fraction of total inorganic sediment supplied to the Klamath River on an annual basis originates from the upper and middle Klamath River (i.e., from Keno Dam to the Shasta River) (Stillwater Sciences 2010) and beneficial uses in the upper Klamath River are currently not impaired due to inorganic suspended material (see Section 3.2, Table 3.2-8).

During the phytoplankton[1] growth season (May through October), organic suspended sediments exhibit a general decreasing trend from upstream to downstream in the Hydroelectric Reach, although the relative decrease through this reach is less than that occurring further upstream where phytoplankton blooms (also called algal blooms) originating in Upper Klamath Lake largely settle out of the water column (Figure C-13; PacifiCorp 2004a; Raymond 2008, 2009, 2010). The most significant decrease (approximately 50 percent) in organic suspended sediments (as TSS) typically occurs between the mouth of Link River and Keno Dam. Further decreases in concentrations of organic suspended sediments can occur in the upstream end of the Hydroelectric Reach, which may be due to the mechanical breakdown of phytoplankton remains and sorting of progressively smaller sizes of natural organic matter (NOM) in the turbulent river reaches between J.C. Boyle Dam and Copco No. 1 Reservoir, as well as by dilution from the springs immediately downstream from J.C. Boyle Dam. By the upstream end of Copco No. 1 Reservoir, average TSS concentrations are approximately 70 percent lower

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[1] Microscopic organisms, including algae, bacteria, protists, and other single-celled plants, that float in the water column of fresh and salt waters and obtain energy from photosynthesis.

than those measured at the mouth of Link River (Figure C-13; PacifiCorp 2004a; Raymond 2008, 2009, 2010).
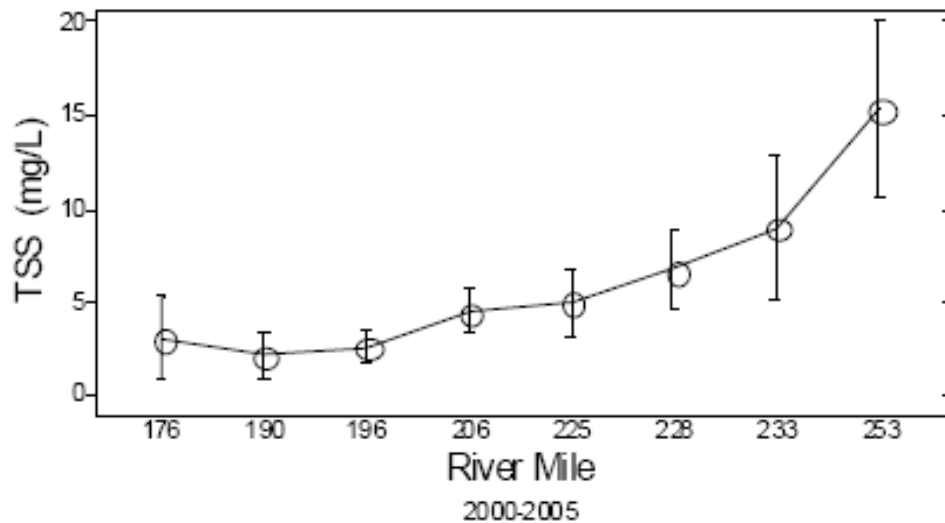
Figure C-13. Mean May to October Total Suspended Solids (TSS) Values for Data Collected from Various Sites in the Klamath River Between 2000 and 2005. Error bars depict 90 percent confidence interval of the mean. The location of features in the graph are specified using 2008 river mile designations, which are slightly different from updated 2018 river miles (see Table 3.2-1). The location of key landmarks in 2008 river mile designations: RM 176 = Klamath River at I-5 crossing; RM 190 = Klamath River downstream of Iron Gate Dam; RM 196 = Klamath River downstream of Copco No. 2 Powerhouse; RM 206 = Klamath River upstream of Copco No. 1 Reservoir; RM 220 = Klamath River downstream of J.C. Boyle Powerhouse; RM 225 = Klamath River downstream of J.C. Boyle Dam; RM 228 = Klamath River upstream of J.C. Boyle Reservoir; RM 233 = Klamath River downstream of Keno Dam; RM 253 = mouth of Link River. Source: Raymond 2008.

Despite the mechanisms supporting decreased longitudinal concentrations of organic suspended sediments in the riverine portions of the Hydroelectric Reach, concentrations in this reach can also increase due to large seasonal phytoplankton blooms occurring in Copco No. 1 and Iron Gate reservoirs. TSS values in Copco No. 1 Reservoir during the phytoplankton growth season (May through October) typically range less than 2 to 20 mg/L and those in Iron Gate Reservoir range less than 2 to 14 mg/L, although intense phytoplankton blooms can result in TSS levels greater than 20 mg/L (Raymond 2008, 2009, 2010). During 2003 sampling by PacifiCorp, a particularly high TSS measurement of 280 mg/L was recorded in the epilimnion of Copco No. 1 Reservoir during May. Simultaneous measurements of suspended sediments measured in the outflow to the reservoir indicated only 4.8 mg/L TSS (FERC 2007), suggesting that the suspended sediment source (phytoplankton cells) had largely settled out of the water column within the reservoir. Since powerhouse withdrawals for Copco No. 1 and Iron Gate dams are from depths of approximately 9.8 m (32 ft) to 10.7 m (35 ft) below the water surface when the reservoirs are full (Section C.1.1.1), only portions of the extensive phytoplankton blooms positioned closer to the water surface may be transported to the downstream Klamath River. During 2009 water quality monitoring, TSS measured in

J.C. Boyle Reservoir ranged less than 2 to 6.8 mg/L from May through November. Levels in Copco No. 1 and Iron Gate reservoirs levels were somewhat greater, with TSS ranging less than 2 to 9.6 mg/L in Copco No. 1 Reservoir (peak in August) and less than 2 to 7.2 mg/L in Iron Gate Reservoir (peak in May) (Watercourse Engineering, Inc. 2011a). Additional water quality monitoring has been conducted from 2010 to 2015 with TSS data from May through October that generally supports previous findings that TSS in Copco No. 1 Reservoir was higher than TSS in Iron Gate Reservoir. TSS in Copco No. 1 Reservoir from 2010 to 2015 usually ranged from the lower reporting limit (0.5 to 5 mg/L) to 17.6 mg/L, with peak TSS measured at 140 mg/L in 2013 and 72.4 mg/L in 2015. TSS in Iron Gate Reservoir from 2010 to 2015 usually ranged from the lower reporting limit (0.5 to 5 mg/L) to 9.2 mg/L, with a peak TSS of 37.6 mg/L measured in 2014 (Watercourse Engineering, Inc. 2011b, 2012, 2013, 2014, 2015, 2016).

There are currently 13.1 million cubic yards of sediment deposits stored within J.C. Boyle, Copco No. 1, Copco No. 2, and Iron Gate reservoirs (USBR 2012) (see also Table 2.8). Prior estimates of the sediment deposits were 14.5 million cubic yards (Eilers and Gubala 2003) and 20.4 million cubic yards (GEC 2006). Sediment texture analysis results of the current reservoir deposits indicate that the deposits are composed of predominantly fine material (i.e., silt and clay less than 0.0625 mm [GEC 2006]; see also Section 3.11) with 3 to 5 percent of the accumulated material as organic carbon, corroborating interpretation of longitudinal suspended sediment patterns and indicating that in-reservoir and upstream phytoplankton growth is largely intercepted and retained in reservoir sediments in the Hydroelectric Reach.

C.2.2   Mid- and Lower Klamath Basin

C.2.2.1   Iron Gate Dam to Salmon River

Immediately downstream from Iron Gate Dam (RM 193.1), inorganic suspended sediment tend to increase with distance downstream from the dam during winter months. Two of the three tributaries that contribute the largest amount of sediment to the Klamath River on an annual basis are in this reach. The Scott River, which enters the mainstem Klamath River at RM 145.1, contributes 607,300 tons per year of suspended sediment or 10 percent of the cumulative average annual delivery from the basin. The Salmon River (RM 66.3) contributes 320,600 tons per year or 5.5 percent of the cumulative average annual delivery from the basin (Stillwater Sciences 2010).

During the phytoplankton growth season (May to October), suspended sediments immediately downstream from Iron Gate Dam are relatively lower than upstream locations, with generally low (less than 5 to 8 mg/L) concentrations for 2000 to 2005 (PacifiCorp 2004a; Raymond 2008, 2009, 2010) (Figure C-13). Between Iron Gate Dam and Seiad Valley (RM 132.7), suspended sediments can increase; for example TSS concentrations near the Shasta River confluence (RM 179.5) for the period 2000 to 2005 were roughly 1 mg/L greater than those measured further upstream at Iron Gate Dam (Figure C-13), and during 2009 monitoring, TSS ranged 0.87 to 4.4 mg/L downstream from Iron Gate Dam (RM 193.1), increasing to 2.5 to 11.5 mg/L downstream from Seiad Valley (RM 132.7) (Watercourse Engineering, Inc. 2011a)[2]. Additional TSS monitoring in the Klamath River was conducted from 2010 to 2015 with TSS in 2015 ranging from 0.8 to 3.8 mg/L downstream of Iron Gate Dam to 4.2 to 155.0 mg/L downstream of Seiad

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[2] This data set includes measurements in November and December 2009 as well.

Valley (Watercourse Engineering, Inc. 2011b, 2012, 2013, 2014, 2015, 2016). In 2015, TSS in the Klamath River downstream of Seiad Valley peaked at 155.0 mg/L in July, but otherwise did not exceed 13.0 mg/L between May and October (Watercourse Engineering, Inc. 2016). The pattern of suspended sediments increasing in the Klamath River downstream from Seiad Valley compared to downstream of Iron Gate Dam may be related to the transport of some portion of the in-reservoir phytoplankton blooms to downstream reaches of Klamath River. River bed scour may also cause resuspension of previously settled materials and increases in summer and fall TSS from 0 to 20 miles downstream from Iron Gate Dam (Figure C-14). Farther downstream, near the confluence with the Scott River (RM 145.1 or approximately 47 miles downstream from Iron Gate Dam) concentration of suspended sediments tend to decrease with distance as suspended sediment gradually settle out of the water column or are diluted by tributary inputs (Armstrong and Ward 2008). Chlorophyll-*a* data show a similar trend (see Section C.6.2.1).
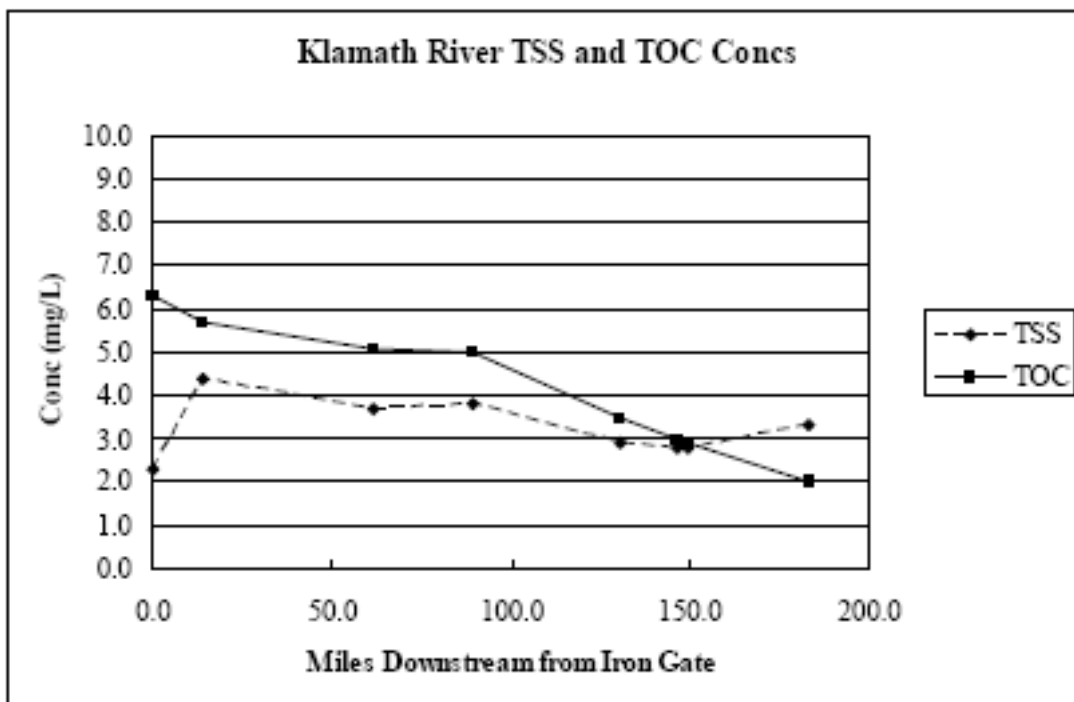
Figure C-14. Average TSS and Total Organic Carbon in the Klamath River Downstream from Iron Gate Dam during June to October 2001 to 2005. River Miles Specified Are Based on the River Miles Designations Used in 2008 and Differ Slightly from 2018 River Mile Designations. Source: Armstrong and Ward 2008.

C.2.2.2   Salmon River to Klamath River Estuary

As in other reaches of the Klamath River, seasonal variation in turbidity and suspended sediments is evident in the Klamath River from the Salmon River (RM 66.3) to the Klamath River Estuary (RM 0 to 3.9), with peak summer turbidity values associated with organic matter (i.e., algal blooms) and peak spring and winter turbidity values associated with inorganic sediments that are mobilized during high flow events (Stillwater Sciences 2009).

Historical (1950 to 1979) suspended sediment data (in SCC) for the Klamath River at Orleans (RM 58.9) (USGS gage no.11523000) range from less than 5 mg/L during summer (low-flow) periods to greater than 5,000 mg/L during winter (high-flow) periods, although some high (>1,000 mg/L) suspended sediment events have occurred during summer months (e.g., 1974, see Figure C-15). During the winter periods, elevated suspended sediment levels are typically associated with storm events and high flows, lasting on the order of days to weeks. More recent data indicate that suspended material levels in the lower Klamath River from the Salmon River confluence (RM 66.3) to the Estuary (RM 0 to 3.9) can be similar to those measured in the upstream reach from Iron Gate Dam to the Salmon River (RM 66.3). During 2009 monitoring, TSS values measured at Orleans were generally 1.1 to 13.3 mg/L between May and December, with peak values (approximately 56 mg/L) occurring during October (Watercourse Engineering, Inc. 2011a). TSS measured at Orleans from 2010 to 2015 had similar trends with TSS usually between 1.2 to 17 mg/L, but TSS occasionally peaking anywhere from 71 mg/L in August 2014 up to 437 mg/L in June 2010 (Watercourse Engineering, Inc. 2011b, 2012, 2013, 2014, 2015, 2016).

Results from grab samples collected by the Yurok Tribe Environmental Program during the period 2003 to 2004 indicate that TSS ranged less than 1.0 to 3.2 mg/L upstream of the Trinity River (RM 43.3) and less than 1.0 to 14.0 farther downstream at Turwar (RM 5.6), with the peak value (14.0 mg/L) occurring in December 2003 (YTEP 2005). However, the majority of the grab samples were collected from June to September and only two grab samples were collected in December and January. The data exhibit similar values for 2007, with the highest TSS (up to 16.0 mg/L) observed at Turwar in September of that year (Fetcho 2008). Additional water quality monitoring has been conducted by the Hoopa Valley Tribal Environmental Protection Agency (HVTEPA) from 2008 to 2012 and the Yurok Tribe Environmental Program from 2006 to 2014 with TSS at sites from upstream of the confluence with the Trinity River to the Klamath River Estuary (Sinnott 2008, 2009a, 2009b, 2010b, 2011b, 2012b; Hanington and Torso 2013; HVTEPA 2013; Hanington and Stawasz 2014; Hanington and Cooper-Carouseli 2014). Between 2008 and 2012, TSS in the Klamath River at Saints Rest Bar (RM 44.9) ranged from 1.0 to 15 mg/L, with TSS reaching a peak of 33.0 mg/L in October 2009. In 2014, TSS in the Klamath River at Weitchpec, upstream from the Trinity River (RM 43.6), ranged from 1 to 12 mg/L while TSS in the Klamath River at the Turwar Boat Ramp (RM 6) ranged 1 to 20 mg/L with the peak values occurring in February at Weitchpec and December at the Turwar Boat Ramp (Figure C-16; Hanington and Cooper-Carouseli 2014).

Figure C-15. Suspended Sediment (mg/L in SCC) Grab Samples for USGS Klamath River at Orleans (USGS Gage No. 11523000) (RM 58.9) 1950 to 1979. Source: USGS 2011 (http://waterdata.usgs.gov/nwis)

Figure C-16. Suspended Sediment (mg/L TSS) Grab Samples at the Lower Estuary Surface (LES [RM 0]), the Klamath River at Turwar Boat Ramp (TG [RM 6]), the Klamath River Upstream of Tully Creek (TC [RM 40.1]), the Klamath River at Weitchpec (Upstream of the Trinity River) (WE [RM 43.6]), and the Trinity River Upstream of the Confluence with the Klamath River (TR). Source: Hanington and Cooper-Carouseli 2014.

The Trinity River contributes 3,317,300 tons of sediment per year to the lower Klamath River or 57 percent of the cumulative average annual delivery from the basin (Stillwater Sciences 2010). Mass wasting, bank erosion, and other natural erosion processes contribute a large but currently unknown portion of the total fine sediment supply to the lower Klamath River, along with management activities such as timber harvest and road construction along tributaries (USDA Forest Service 2004; Stillwater Sciences 2010). When combined with the steep terrain, granular soil matrix, and high precipitation, these sources may be a primary contributor to fine sediment deposits found in deep pools near cultural sites in the lower Klamath River (FERC 2007).

Available historical (1958 to 1996) suspended sediment data for the Klamath River at Klamath Glen (RM 5.9) (USGS gage no. 11530500) indicates values of less than 5 mg/L (in SCC) during summer (low-flow) periods to greater than 500 mg/L during winter (high-flow) periods, although one high (>750 mg/L) suspended sediment event appears to have occurred during the early fall (i.e., October 1977, see Figure C-17).
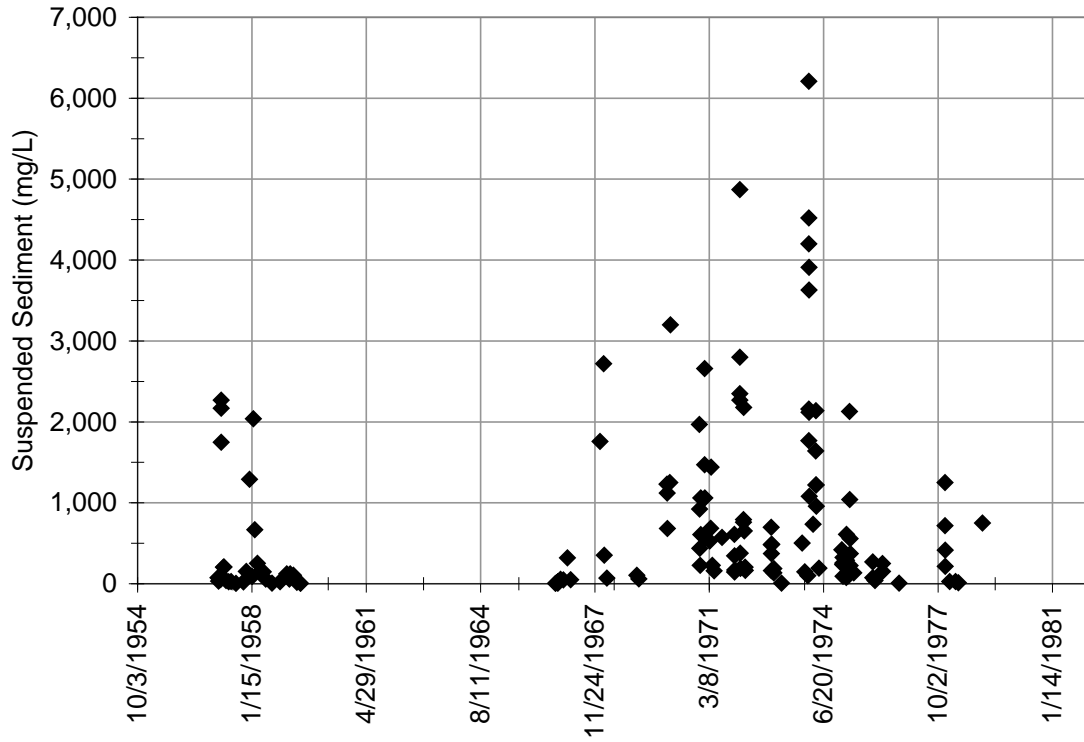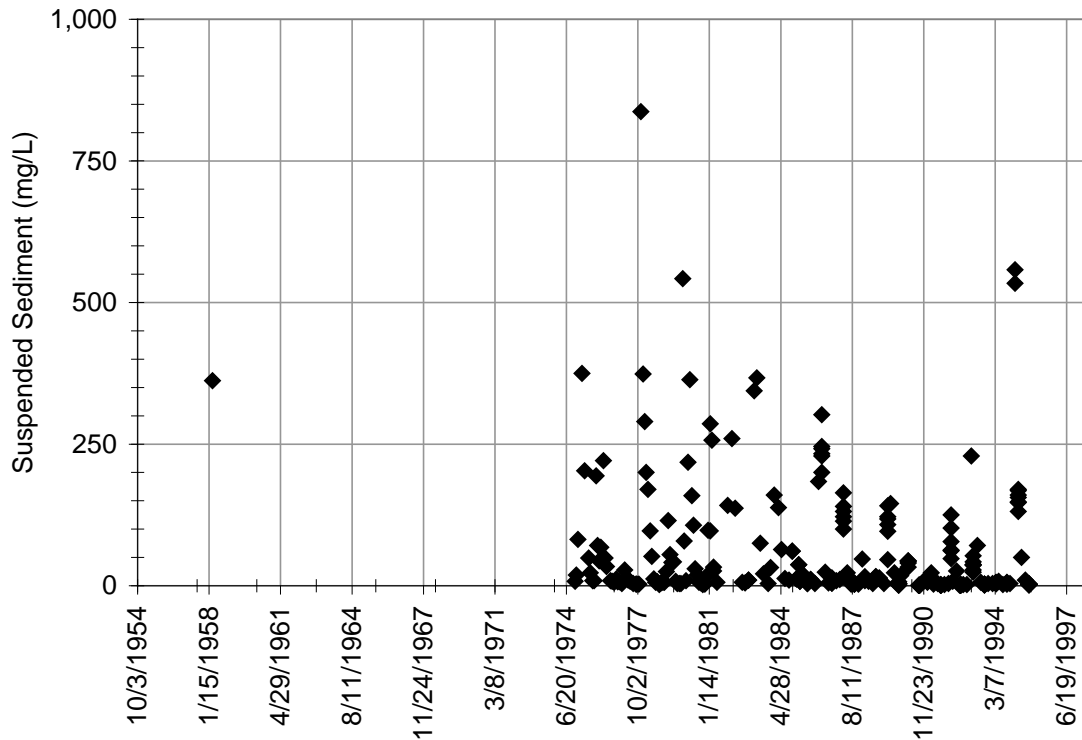
Figure C-17. Suspended Sediment (mg/L) Grab Samples for USGS Klamath River near Klamath (USGS Gage No. 11530500) (RM 7) 1958 to 1995. Source: USGS 2011 (http://waterdata.usgs.gov/nwis).
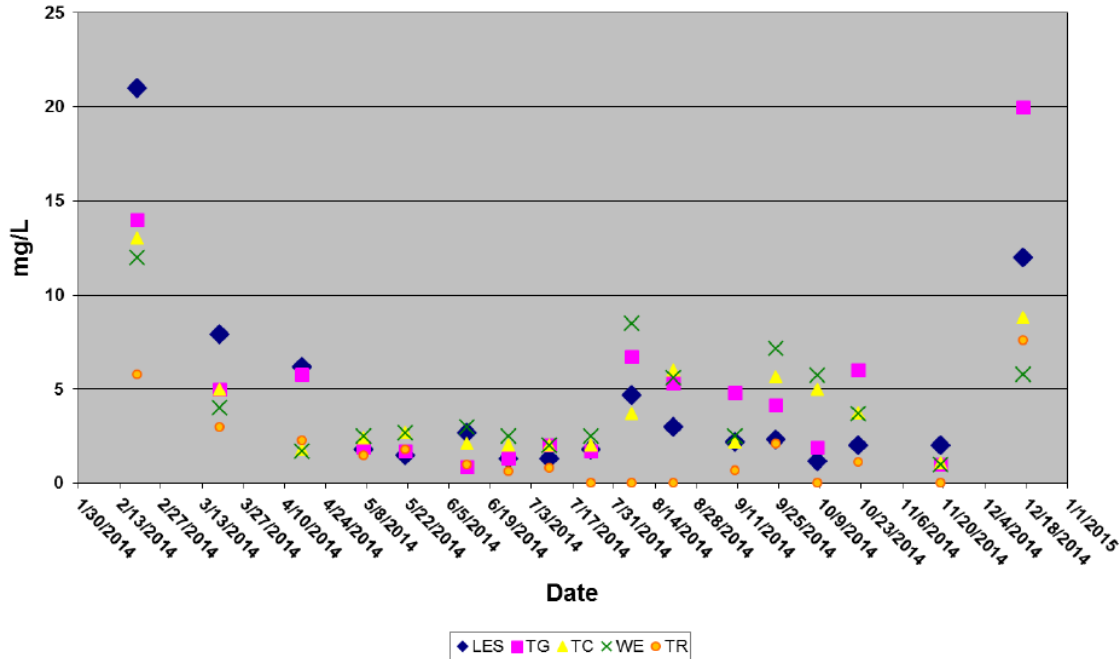
C.2.2.3   Klamath River Estuary

An analysis of collected TSS data in the Klamath River Estuary indicates that TSS are variable but generally similar to those measured at upstream sites in the lower Klamath River (YTEP 2004, 2005; Sinnott 2008). For 2003 to 2004, TSS levels were less than 1.0 to 3.2 mg/L for surface waters in the mid- and lower-estuary, and slightly greater (1.8 to 10.0 mg/L) at depth (YTEP 2004, 2005). During May to December 2009, measured TSS levels were generally 2.1 to 12.7 mg/L, with the peak value (17.9 mg/L) occurring in May (Watercourse Engineering, Inc. 2011a). More contemporary data measured between 2006 to 2014 show a larger range in the TSS in the lower Klamath River Estuary, with TSS ranging from 1.3 to 21 mg/L in 2014[3] (Figure C-16; Sinnott 2008, 2009a, 2009b, 2010b, 2011b, 2012b; Hanington and Torso 2013; Hanington and Stawasz 2014; Hanington and Cooper-Carouseli 2014). Turbidity measurements in small tributaries (e.g., McGarvey, Den, Blue, and Turwar creeks) immediately upstream or within a few river miles upstream of the Klamath River Estuary exhibit peak values during winter high flow periods (i.e., storm events), with measured values exceeding 500 NTU during December through February 2004 (YTEP 2005). Additional turbidity measurements at sites from upstream of the confluence with the Trinity River to the

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[3] The various reports cited for available TSS information provide data for each individual year. There is currently no synthesis report for TSS monitoring data for the Klamath River. The most recent data range that highlights the intent of the discussion is presented, along with multiple citations to available reports.

Klamath River Estuary have been collected by the Yurok Tribe Environmental Program from 2010 to 2014 with peak turbidity typically occurring between December and April and a secondary smaller peak in turbidity occurring in the summer or fall (Sinnott 2008, 2009a, 2009b, 2010b, 2011b, 2012b; Hanington and Torso 2013; Hanington and Stawasz 2014; Hanington and Cooper-Carouseli 2014). In 2014, turbidity in the Klamath River Estuary ranged from 0.49 to 9.4 NTU with the peak occurring in February (Figure C-18; Hanington and Cooper-Carouseli 2014). During late spring through early fall, when average rates of precipitation in the Klamath Basin are relatively lower, inorganic suspended sediments and turbidity in the Klamath River Estuary are generally lower as well.

Blooms of phytoplankton within and upstream of the Klamath River Estuary have the potential to cause large spikes in turbidity and organic suspended sediments in the estuary. This occurred during the extensive phytoplankton bloom detected throughout at least 40 river miles of the lower Klamath River in September 2007 (Kann 2007a, b, c, d). In the lower estuary, increases in nutrient levels and phytoplankton concentrations were correlated with an increase in TSS from 2.2 mg/L on August 21, 2007 to 9.0 mg/L on September 18, 2007, and increases in nutrients, phytoplankton levels, and TSS during that period were measured as far upstream as Iron Gate and Copco No. 1 reservoirs (Asarian et al. 2009). Thus, the observed 2007 increase in estuarine TSS may have been influenced by phytoplankton growth originating in Iron Gate and Copco No. 1 reservoirs. The downstream transport of phytoplankton from Iron Gate and Copco No. 1 reservoirs into the Middle and Lower Klamath River and the Klamath River Estuary is discussed more in Section C.6 and Section 3.4 *Phytoplankton and Periphyton.*
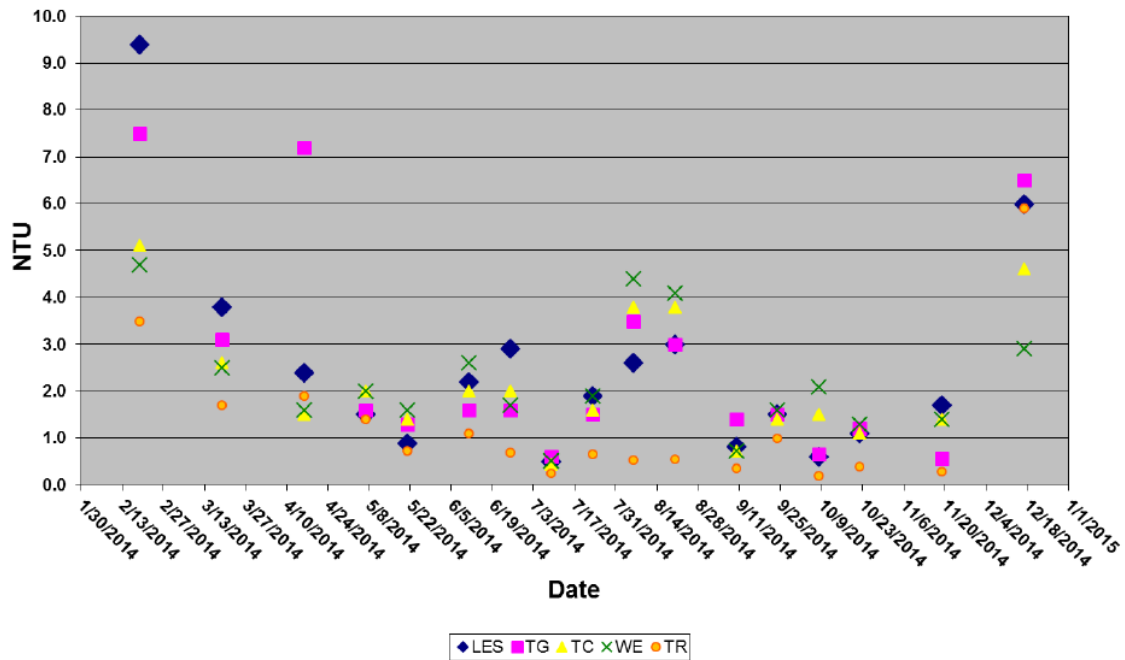

Figure C-18. Turbidity (NTU) Grab Samples at the Lower Estuary Surface (LES [RM 0]), the Klamath River at Turwar Boat Ramp (TG [RM 6]), the Klamath River Upstream of Tully Creek (TC [RM 40.1]), the Klamath River at Weitchpec (Upstream of Trinity River)(WE [RM 43.6]), and the Trinity River Upstream the Confluence with the Klamath River (TR). Source: Hanington and Cooper-Carouseli 2014.

C.3   Nutrients

Nutrients are critical for the support of primary productivity (i.e., plant growth) in both terrestrial and aquatic ecosystems. High levels of nutrients (nitrogen and phosphorus) in lakes and rivers have the potential to impact overall water quality by increasing rates of phytoplankton growth and decay, which can lead to increased levels of turbidity, large fluctuations in dissolved oxygen concentrations and pH levels, as well as potential increases of toxic substances such as ammonia (NH_4^+/NH_3), hydrogen sulfide (H_2S), and release of heavy metals from low oxidation-reduction potential at the sediment water interface (see Section 3.2.3.1 for additional background information on water quality processes in the Klamath Basin). Dissolved nutrients (e.g., ortho-phosphorus, nitrate, and ammonium) can be used directly by phytoplankton, whereas particulate nutrients (e.g., organic phosphorus, organic nitrogen) are not readily bioavailable for most phytoplankton species.

C.3.1   Upper Klamath Basin

C.3.1.1   Hydroelectric Reach

Nutrients are introduced into the mainstem Klamath River Area of Analysis primarily by the Upper Klamath Lake which inputs nitrogen and phosphorus (Kann and Walker 1999, ODEQ 2002, PacifiCorp 2004b, Deas and Vaughn 2006, FERC 2007, Sullivan et al. 2008, Asarian et al. 2010) and the Lost River Basin via the Klamath Straits Drain and the Lost River Diversion channel which inputs nutrients and organic matter (Lytle 2000, Mayer 2005, Sullivan et al. 2009, Sullivan et al. 2011, Kirk et al. 2010). Historical and contemporary nutrient data indicate that, on an annual basis, nutrients in the Hydroelectric Reach tend to be lower than upstream due in part to dilution from springs downstream of J.C. Boyle Reservoir. According to Asarian et al. (2010), who cited analysis by IFR and PCFFA (2009), Gard (2006), and TetraTech (2009), the long-term average nutrient concentrations of the springs downstream of J.C. Boyle Dam was estimated to be on the order of 0.2 mg/L for total nitrogen (TN) and on the order of 0.07 mg/L for total phosphorus (TP) using mixing equations and PacifiCorp's 2001-2007 nutrient sampling data. The Klamath TMDL model TN and TP concentrations, derived from model calibration through 2000, are consistent with these values.

Total Nitrogen and Total Phosphorus Patterns
The settling of particulate matter and associated nutrients in the larger Lower Klamath Project reservoirs contributes significantly to decreases in nutrient concentrations in the Klamath River from the Oregon-California state line to Iron Gate Dam (PacifiCorp 2004a, FERC 2007, Butcher 2008, Asarian et al. 2009, Asarian and Kann 2011, Oliver et al. 2014). In J.C. Boyle Reservoir (RM 229.8), the furthest upstream reservoir in the Hydroelectric Reach, concentrations of TN and TP measured between the inflow and outflow are typically similar, likely due to the shallow depth and short residence time characteristic of this impoundment (PacifiCorp 2006), indicating that relatively little nutrient retention occurs in this reservoir. A decreasing longitudinal trend in TN and TP from downstream of J.C. Boyle to Iron Gate Dam is generally consistent in analyses of datasets from 2005 to 2015, but the annual range and seasonal variations of TN and TP concentrations occasionally results in TN and TP plateauing or slightly increasing in the Copco No. 1 to Iron Gate portion. Analysis of nutrient data from 2005 to 2008 found the flow-weighted longitudinal TN and TP concentrations generally decrease with distance downstream from J.C. Boyle Dam with a strongly downward trend through Copco No. 1 and Iron Gate reservoirs, particularly for TN, due to nutrient retention in Copco No. 1 and

Iron Gate reservoirs (see Figure C-19 and C-20; Asarian et al. 2009; Asarian et al. 2010). Nutrient data from 2007 to 2015 supports the overall annual longitudinal trend of TN and TP, but also reveals the longitudinal variability in both the range and median TN and TP concentrations through the Hydroelectric Reach (Raymond 2008, 2009, 2010; Watercourse Engineering, Inc. 2011a, 2011b, 2012, 2013, 2014, 2015, 2016). Data from 2012 to 2015 highlight the range of TN and TP variations downstream from J.C. Boyle Dam (Figures C-21 and C-22; Watercourse Engineering, Inc. 2013, 2014, 2015, 2016). Asarian and Kann (2011) observe an overall longitudinal decrease in TN through the Hydroelectric Reach of the Upper Klamath River with concentrations usually highest upstream of Copco No. 1, intermediate upstream of Iron Gate, and lowest downstream of Iron Gate Dam. Longitudinal trends in TP show a seasonal variation though with TP generally decreasing through the Hydroelectric Reach from January to May and from July to August/September (varies by year), but TP exhibiting an increasing trend from August/September until approximately December (Asarian and Kann 2011). Oliver et al. (2014) also note TN and TP usually decrease with distance downstream from J.C. Boyle, but TP is found to occasionally slightly increase from J.C. Boyle to Iron Gate primarily between July and November.

Annual data from 2000 through 2004 and early modeling studies by PacifiCorp conducted for the FERC relicensing process indicates that Copco No. 1 and Iron Gate reservoirs act primarily as TN and TP sinks due to trapping of phytoplankton detritus (PacifiCorp 2004a; FERC 2007), but subsequent analyses found that while overall annual retention is likely occurring, the Lower Klamath Project reservoirs can also serve as seasonal sources of TN and TP (though far less for TN than for TP) through the release of nutrients from reservoir sediments into the water column during periods of phytoplankton decomposition and seasonal hypolimnetic anoxia, and possibly through direct nitrogen fixation from the atmosphere by cyanobacteria [blue-green algae] (Kann and Asarian 2005, 2006; Asarian and Kann 2006a, 2006b; Butcher 2008; Asarian et al. 2009; Asarian et al. 2010; Asarian and Kann 2011; Oliver et al. 2014). Seasonal variations of TN and TP occurred from 2005 to 2008 due to release of nutrients to the water column during periods of seasonal hypolimnetic anoxia (Figures C-23 and C-24; Asarian et al. 2010). Asarian and Kann (2011) detail the 2005-2010 average monthly TN and TP variations at various depths in Copco No. 1 and Iron Gate reservoirs. TN concentrations generally are highest at the deepest depths in both reservoirs except in July to September when organic N was very high during phytoplankton blooms (also called algal blooms). Asarian and Kann (2011) report TP concentrations are approximately uniform with depth when the reservoirs are not stratified, but the TP increases significantly with depth, especially in the bottom layer, during the period the reservoirs stratify (June to October in Copco No. 1 and June through November in Iron Gate). Data presented in Asarian et al. (2009), Asarian et al. (2010), and Asarian and Kann (2011) suggest that much of the TP released from sediments in Copco No. 1 and Iron Gate reservoirs during summertime anoxia remains in the hypolimnion until the reservoirs begin to turn over in the fall, rather than being released to downstream river reaches during the summer period of peak periphyton growth. A frequent and notable exception occurs during August to November, when TP concentrations are often higher in Iron Gate Reservoir than they are upstream of Copco No. 1 Reservoir. This is likely due to the combination of internally-driven nutrient dynamics related to algal bloom crashes in Copco No. 1 and Iron Gate reservoirs and an approximately 1- to 2-month temporal lag due to the longer hydraulic retention time of the reservoirs as compared to free-flowing river reaches (Kann and Asarian 2007; Asarian et al. 2009; Asarian et al. 2010; Asarian and Kann 2011; Watercourse Engineering, Inc. 2011a). Oliver et al.

(2014) further explore the seasonality of TP dynamics with algal bloom variations highlighting measurements that show TP concentrations are lowest during May to July ("bloom period"), highest during August to October ("post-bloom period"), and then decrease during the winter periods. These results indicate that some release of TP may occur at times which could stimulate downstream periphyton growth.
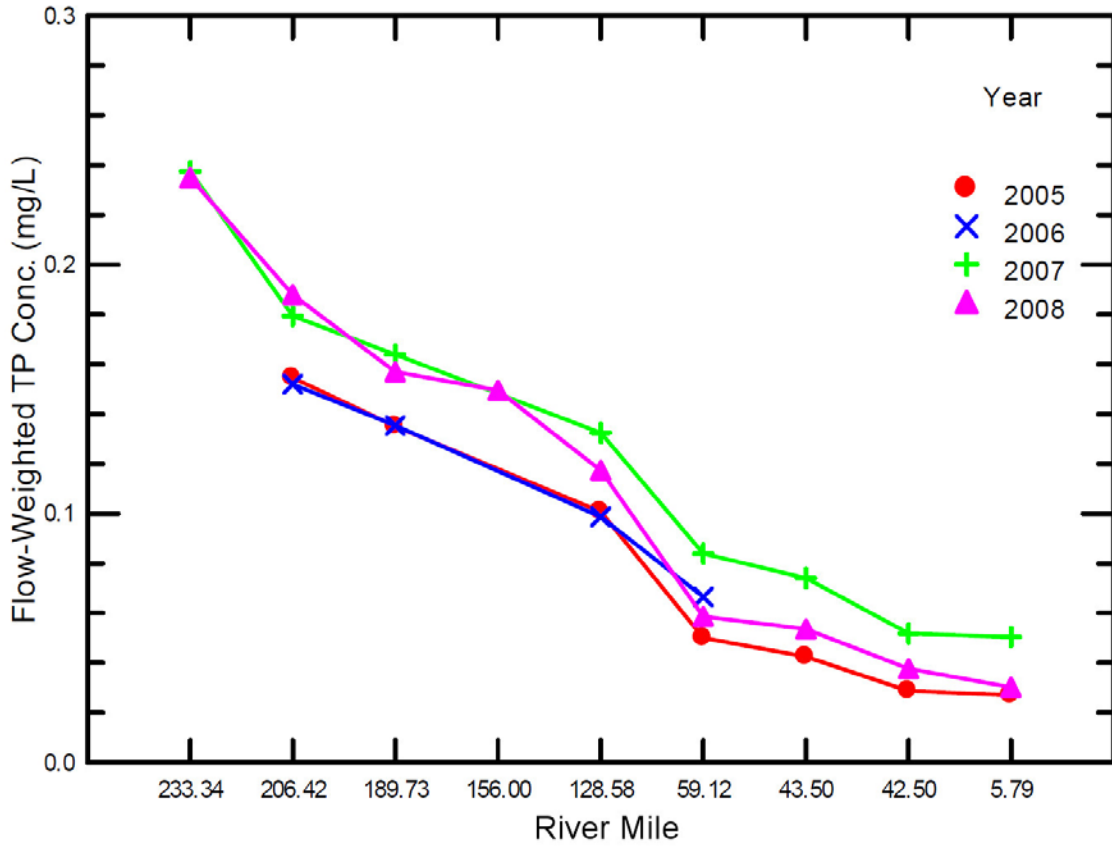
Figure C-19. Summary of Flow-Weighted Mean Concentration (mg/L) for TP at Mainstem Klamath River Sites for the Months of June to October (2005 to 2008). River Miles Specified Are Based on the River Miles Designations Used in 2010 and Differ Slightly from 2018 River Mile Designations (Table 3.2-1). Location of Mainstem Klamath River Measurements in 2010 River Mile Designations: Downstream Keno Dam (RM 233.34), Upstream Copco No. 1 Reservoir (RM 206.42), Downstream Iron Gate Dam (RM 189.73), Walker Bridge (RM 156.00), Seiad Valley (RM 128.58), Orleans (RM 59.12), Upstream Trinity River Confluence (RM 43.50), Downstream Trinity River Confluence (RM 42.50), Turwar (RM 5.79). Source: Asarian et al. 2010.
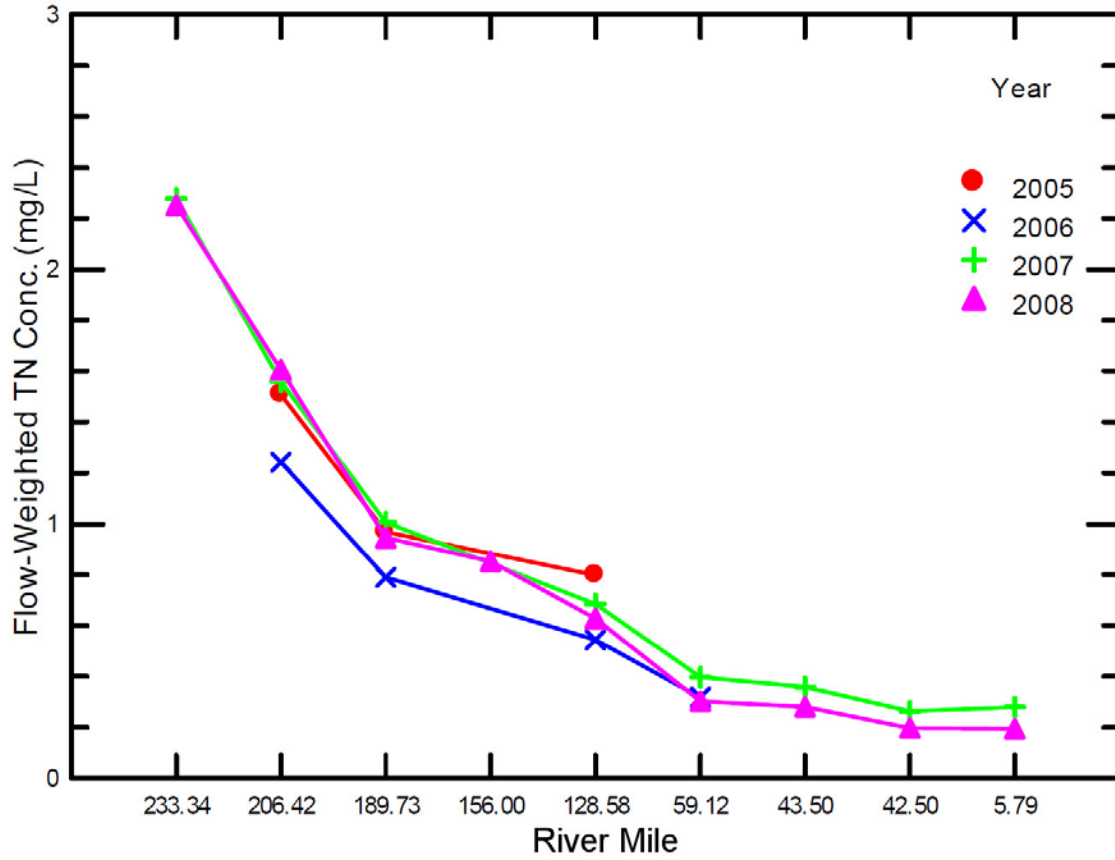
Figure C-20. Summary of Flow-Weighted Mean Concentration (mg/L) for TN at Mainstem
Klamath River Sites for the Months of June to October (2005 to 2008). River
Miles Specified Are Based on the River Miles Designations Used in 2010 and Differ
Slightly from 2018 River Mile Designations (Table 3.2-1). Location of Mainstem
Klamath River Measurements in 2010 River Mile Designations: Downstream Keno
Dam (RM 233.34), Upstream Copco No. 1 Reservoir (RM 206.42), Downstream Iron
Gate Dam (RM 189.73), Walker Bridge (RM 156.00), Seiad Valley (RM 128.58),
Orleans (RM 59.12), Upstream Trinity River Confluence (RM 43.50), Downstream
Trinity River Confluence (RM 42.50), Turwar (RM 5.79). Source: Asarian et al.
2010.
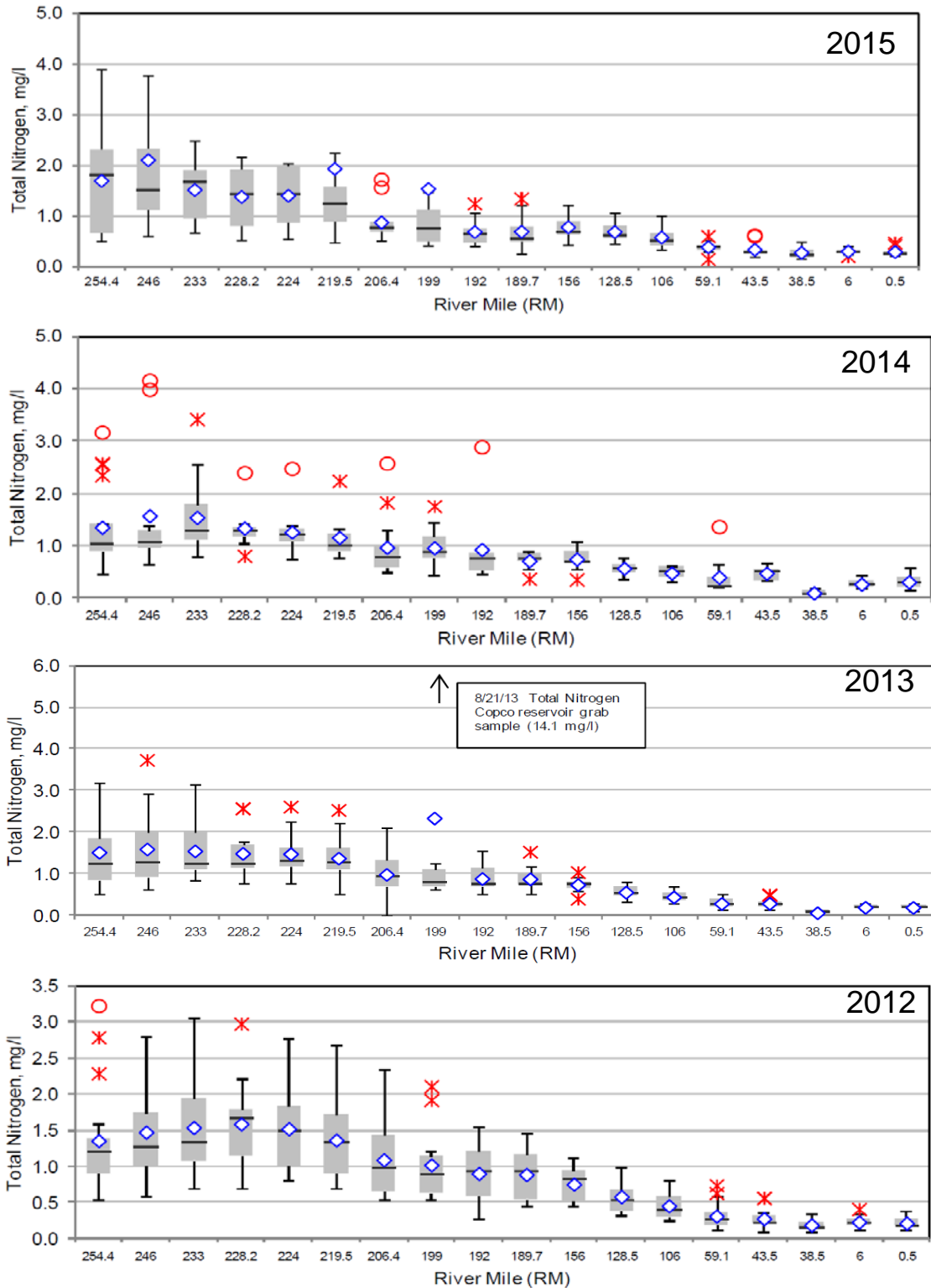
Figure C-21. Klamath River total nitrogen trends including Copco Reservoir (RM 198.7) and
Iron Gate Reservoir (RM 189.7) with median (-), mean (◊), outlier (*), and
extreme outliers (○) identified (Watercourse Engineering, Inc. 2013, 2014, 2015,
2016). River miles specified are based on those accurate at time of reports and
differ slightly from 2018 river mile designations (Table 3.2-1).
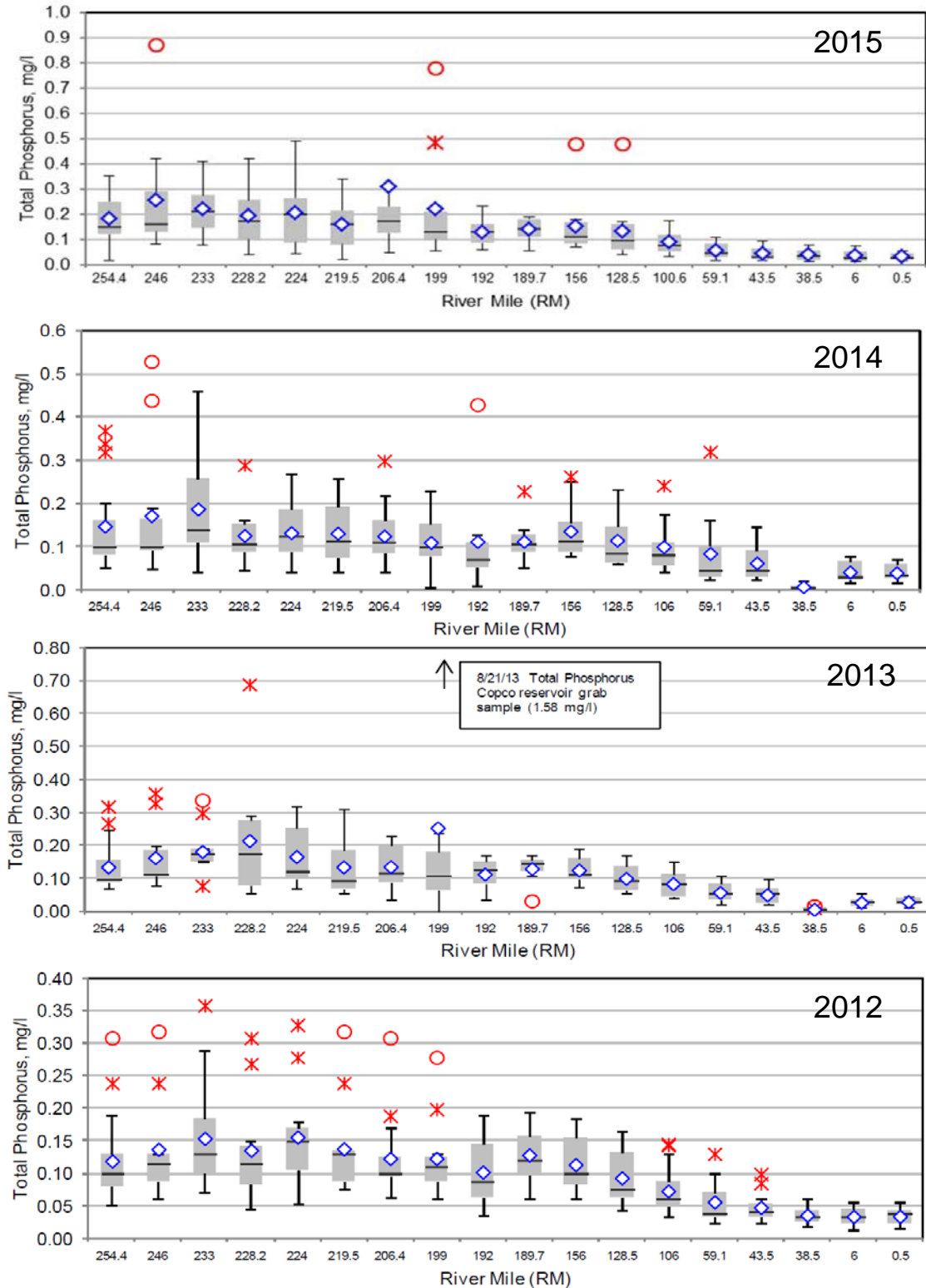
Figure C-22. Klamath River total phosphorus trends including Copco Reservoir (RM 198.7) and Iron Gate Reservoir (RM 189.7) with median (-), mean (◊), outlier (*), and extreme outliers (○) identified (Watercourse Engineering, Inc. 2013, 2014, 2015, 2016). River miles specified are based on those accurate at the time of the reports and differ slightly from 2018 river mile designations (Table 3.2-1).
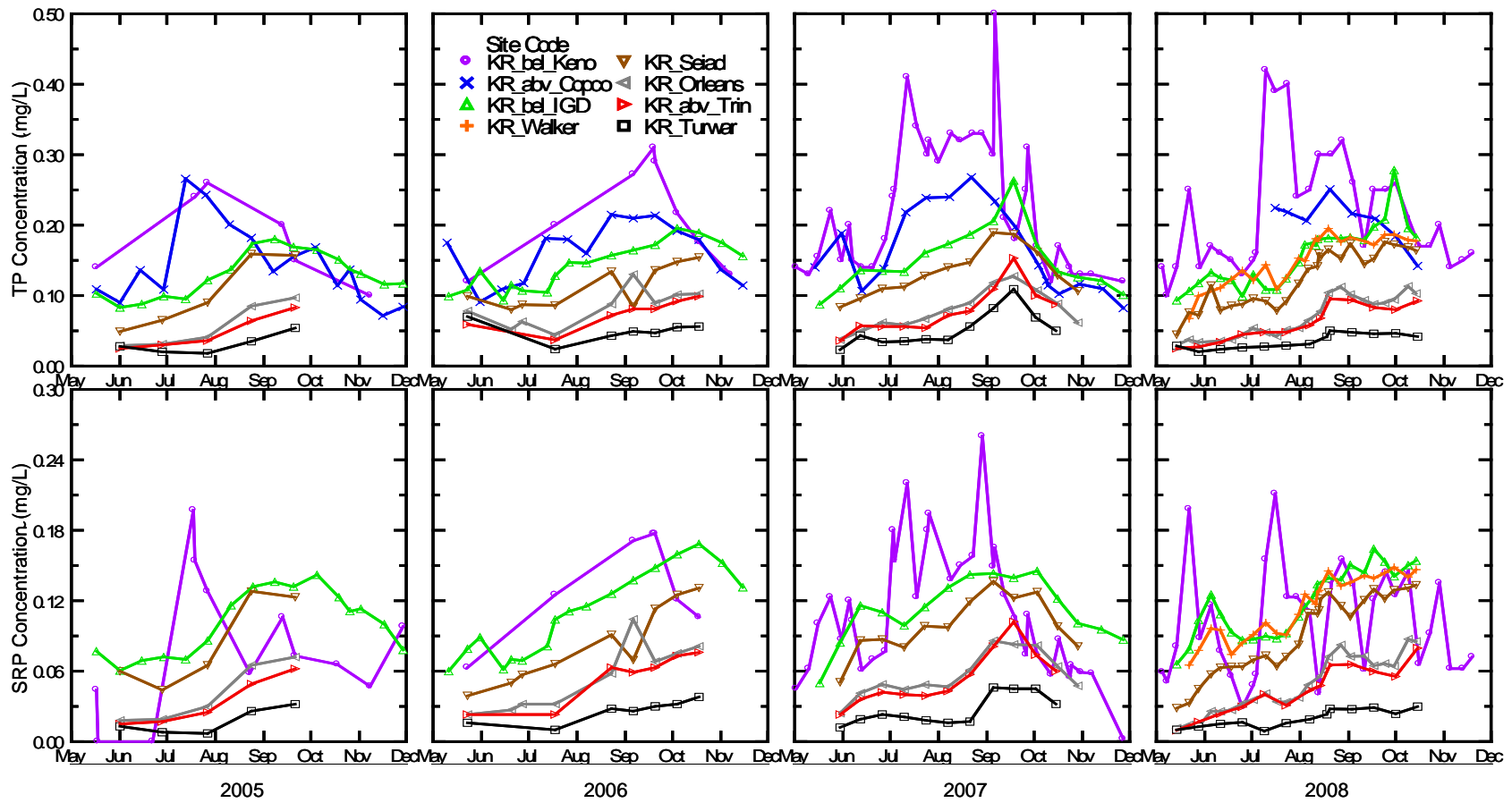
Figure C-23. Time Series of Total Phosphorus (TP) and Soluble Reactive Phosphorus (SRP) Concentrations for Selected Mainstem Klamath River Sites from Downstream Including the J.C. Boyle Dam (RM 229.8) to Turwar (RM 5.6) Reach, May 2005 to November 2008. Source: Asarian et al. 2010.
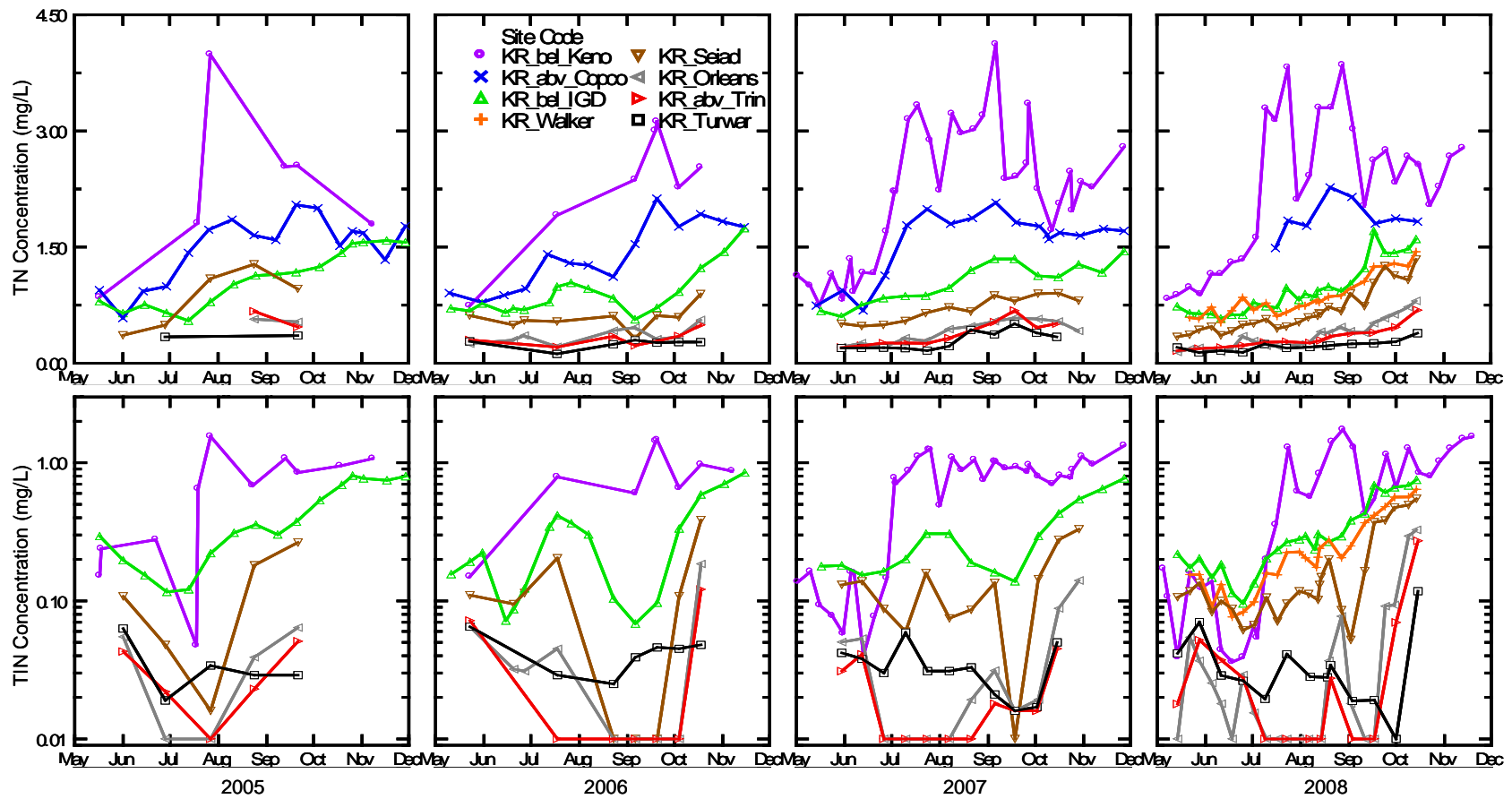
Figure C-24. Time Series of Total Nitrogen (TN) and Total Inorganic Nitrogen (TIN) Concentrations for Selected Mainstem Klamath Sites from Downstream Including the J.C. Boyle Dam (RM 229.8) to Turwar (RM 5.6) Reach, May 2005 to November 2008. Source: Asarian et al. 2010.

With respect to nutrient speciation, internally-driven reservoir nutrient dynamics due to stratification patterns and hydraulic residence time in Iron Gate and Copco No. 1 reservoirs appear to influence ortho-phosphorus and, to a lesser degree, ammonium concentrations within the Hydroelectric Reach.

<u>Orthophosphate</u>
Orthophosphate is a bioavailable and dissolved form of phosphorus that is frequently measured in addition to total phosphorus to assess the nutrients available for uptake by aquatic organisms. Orthophosphate is the most commonly measured form of dissolved phosphorus, but soluble reactive phosphorus (SRP) is also measured and can be used as a surrogate for orthophosphate. SRP concentrations in the riverine portions of the Hydroelectric Reach generally follow a decreasing longitudinal trend through this reach for summer and fall months (i.e., May through November; see Figure C-23). However, concentrations in Iron Gate Reservoir can exceed those of upstream sites (i.e., Klamath River downstream from J.C. Boyle and Copco No. 1 reservoirs) particularly between September and December (Asarian et al. 2009; Asarian et al. 2010; Raymond 2009, 2010; Asarian and Kann 2011; Oliver et al. 2014). Although there are limited data during winter months when the reservoirs are mixed, a synthesis of SRP data from 2005 to 2010 shows concentrations of orthophosphate appear to be more constant throughout the water column, while in stratified periods (i.e., May to October/November) concentrations near the bottom of the reservoirs can reach relatively high levels (Asarian and Kann 2011). For example, orthophosphate concentrations in the bottom waters of Copco No. 1 Reservoir reached 1.4 mg/L in September and October of 2008 and 2009, while surface water concentrations were approximately 0.2 to 0.3 mg/L (see Figure 26 in Raymond [2009] and Figure 22 in Raymond [2010]). Orthophosphorus concentrations in Iron Gate Reservoir during this same period ranged from approximately 0.1 mg/L to 0.3 mg/L (Raymond 2009, 2010). Vertical differences in orthophosphorus concentrations in Iron Gate Reservoir were less than 0.1 mg/L, but the highest concentrations often occurred near the bottom sediments (see Figure 26 in Raymond [2009] and Figure 25 in Raymond [2010]).

<u>Nitrate</u>
Data from 2001 to 2008 indicate that nitrate concentrations often peak in the vicinity of J.C. Boyle Reservoir and decrease through the remainder of the Hydroelectric Reach (Raymond 2009). More recent analyses of nitrate concentrations from 2005 to 2011 in the Hydroelectric Reach support this conclusion and detail seasonal nitrate variations with nitrate substantially higher upstream of Copco No. 1 Reservoir than downstream of Copco No. 1 Reservoir or Iron Gate Reservoir (Asarian and Kann 2011, Oliver et al. 2014). On a seasonal basis, coupled nutrient and phytoplankton data indicate that nitrate levels decrease during algal blooms in the Hydroelectric Reach. Cyanobacteria [blue-green algae] blooms were recorded in Iron Gate and Copco No. 1 reservoirs in summer and fall 2005 coincident with a nitrate decrease of up to 0.8 mg/L between the inflow to Copco No. 1 and the outflow of Iron Gate reservoirs (Kann and Asarian 2007). In 2010 to 2011, Oliver et al. (2014) also observed low nitrate concentrations corresponding to algal blooms measuring nitrate concentrations from 0.01 to 0.08 mg/L NO_3^--N in May-July during the onset and initial peak of the algal bloom, 0.08 to 0.94 mg/L NO_3^--N in August to October following the bloom peak and during the initial bloom decline , 0.39 to 1.01 mg/L NO_3^--N in November to January, and 0.21 to 0.73 mg/L NO_3^--N in February to April. Dilution from the springs downstream from J.C. Boyle Dam also reduces nitrate concentrations in this reach even though the springs are also a relatively constant source of nitrate (Oliver et al. 2014).

<u>Ammonium/Ammonia (NH$_4^+$/NH$_3$)</u>

Ammonium (NH$_4^+$) and ammonia (NH$_3$) are two related forms of nitrogen that influence water quality, with the ratio of ammonium and ammonia dependent on the water temperature and pH (North Coast Regional Board 2010). Ammonium is converted to ammonia at higher water temperatures and higher pH. Ammonium and ammonia are naturally formed in the environment by microbes and some forms of blue-green algae that combine nitrogen from the air with hydrogen (nitrogen fixation), microbes that convert nitrogen in decaying organic matter into ammonia and ammonium, and fish excreting ammonia (USEPA 2013). Ammonium is a nutrient that is directly usable by phytoplankton and as such it can promote phytoplankton growth. At high concentrations, ammonium and ammonia are toxic to aquatic species, with toxicity levels varying by species. Ammonia is more toxic than ammonium (North Coast Regional Board 2010), so it is the form typically measured and reported.

Nutrient data from 2005 to 2010 show ammonia concentrations from May to September are often lowest upstream of Copco No. 1 Reservoir or downstream of Iron Gate Dam with the highest concentrations in Copco No. 1 Reservoir, but ammonia concentrations from October to December are higher upstream of Iron Gate Reservoir and downstream of Iron Gate Dam (Asarian and Kann 2011). Low ammonia concentrations between May and September are attributed to nitrification in the turbulent oxygen-rich Klamath River upstream of Copco Reservoir (Deas 2008). Asarian and Kann (2011) identify deterioration and decay of algal blooms along with reservoir turnover as the likely causes of higher ammonia concentrations from October to December. Seasonal longitudinal trends in the ammonium concentrations in the Hydroelectric Reach show minimums occur during May to July algal bloom periods and maximums occur during November to January (Oliver et al. 2014).

Relatively high levels of ammonium have been recorded in reservoirs in the Hydroelectric Reach especially in the lower reservoir depths. While available data collected to date suggests no actual ammonia toxicity events associated with the operation of the Lower Klamath Project (North Coast Regional Board 2010), elevated ammonia levels in the deeper portions of the hypolimnion of both Copco No. 1 and Iron Gate reservoirs in summer of 2005 exceeded 0.6 mg/L (Figures 12 and 14 in Kann and Asarian 2007), indicating that anoxic conditions are likely causing conversion of organic nitrogen in reservoir deposits to ammonia. From 2001 to 2004, June and November mean ammonia concentrations in Iron Gate Reservoir were 0.1 to 0.3 mg/L to a depth of 45 meters, whereas Copco No. 1 Reservoir concentrations were consistently higher for the 20- to 32-meter depth and were 0.9 to 1.0 mg/L in September and October (FERC 2007). Only minor increases in ammonia (0.05 to 0.1 mg/L) have been observed to occur in the Hydroelectric Reach between upstream of Copco No. 1 Reservoir and downstream from Iron Gate Reservoir, most often during October and November (Kann and Asarian 2005, 2007). The 2005 to 2010 average monthly ammonia concentrations support previous findings showing ammonia concentrations increase substantially (often over an order of magnitude) with depth in Copco No. 1 Reservoir between May and October peaking at over 1.0 mg/L in September/October, while ammonia concentrations in Iron Gate Reservoir increase with depth between May and November usually peaking around 0.4 to 0.8 mg/L in October/November (Asarian and Kann 2011).

C.3.2   Mid- and Lower Klamath Basin

C.3.2.1   Iron Gate Dam to Salmon River

Historical (1950 to 2001) TP data indicate median values of 0.11 to 0.19 mg/L in the Lower Klamath Basin between Iron Gate Dam and Seiad Valley, with the highest values occurring just downstream from the dam (Figure C-25). Variability over the long-term record in this reach is lower than upstream reaches, with concentrations varying from near zero to over 0.3 mg/L for the period of record (Figure C-25). The historical record indicates relatively low variability in orthophosphate concentrations in the reach (as compared with variability in the Upper Klamath Basin), with median values of 0.03 to 0.1 mg/L (PacifiCorp 2004b).
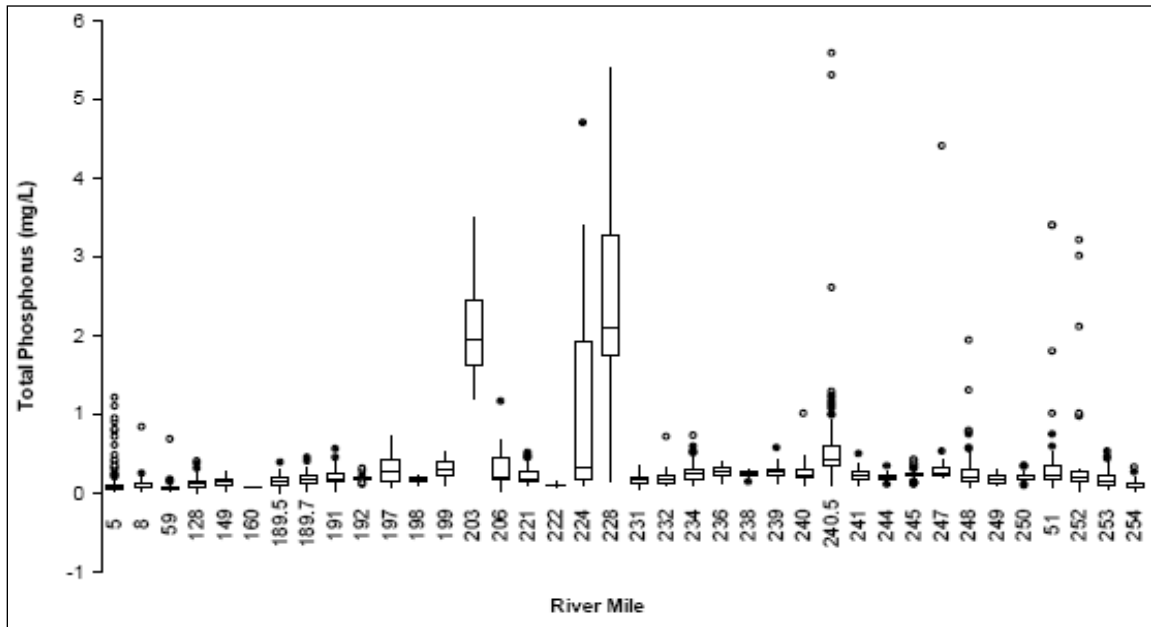
Figure C-25. Box and Whisker Plot of Historical TP Data Collected from Various Sites in the Klamath River from Klamath River at Klamath Glen (RM 5.9) to Klamath River at Link River Dam (RM 259.7) Between 1950 and 2001. Source: PacifiCorp 2004b.

More recent data from 2001 to 2015 also show TP concentrations generally decreasing with distance downstream from Iron Gate Dam and indicate that phosphorus dynamics in the Klamath River immediately downstream from Iron Gate Dam are affected by conditions within the Lower Klamath Project reservoirs (Section C.3.1.1) (Figures C-22 and C-26; Asarian et al. 2010; Asarian and Kann 2013; Oliver et al. 2014; Watercourse Engineering, Inc. 2011a, 2011b, 2012, 2013, 2014, 2015, 2016). Analysis of data from 2001 to 2011 shows the highest TP concentrations occurring during the low flow years of 2001 to 2004 when TP exceeded 0.4 mg/L downstream from Iron Gate Dam and the lowest TP concentrations occurring during the high flow years of 2006, 2010, and 2011 and the moderate flow year of 2005 (Asarian and Kann 2013). TP concentrations vary seasonally with TP peaking after the algal bloom peak and then decreasing during late fall and early winter (Oliver et al. 2014). During May 2005 to November 2008, peak TP concentrations at locations downstream from Iron Gate Dam occurred between mid-August and early October, which is roughly one to two months later than peak timing in upstream reaches and may be due to the hydraulic residence time in Iron Gate and

Copco No. 1 reservoirs, or release of TP from anoxic sediments during summer stratification or following algal bloom and death (Figure C-23). After peaking, the TP concentrations declined steeply in late-fall in some years, but they exhibited a more gradual decline in others (Figure C-23). Orthophosphate or SRP tends to decrease in the mainstem Klamath River with distance downstream from Iron Gate Dam (FERC 2007; Asarian et al. 2010; Asarian and Kann 2013). Seasonal trends in orthophosphate closely follow observed TP concentrations and for the period 2005 to 2008 this phosphorus species regularly accounts for 60 to 90 percent of TP sampled (Asarian et al. 2010; Asarian and Kann 2013; Oliver et al. 2014).
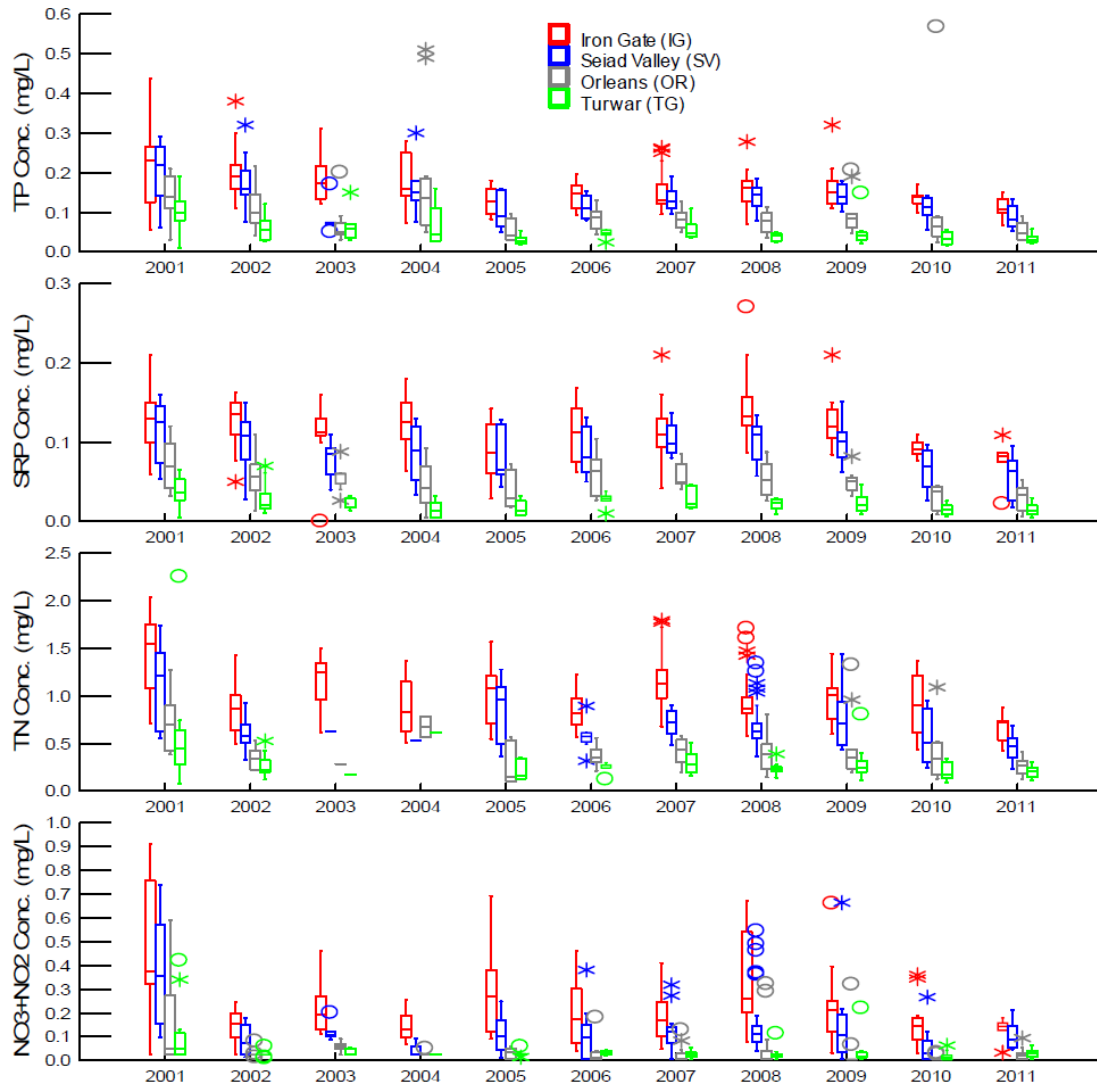
Figure C-26. Box and Whisker Plot of Total Phosphorus (TP), Soluble Reactive Phosphorus (SRP), Total Nitrogen (TN), and Nitrate plus Nitrite Nitrogen (NO3+NO2) for the June-October Period from 2001-2011 at Four Mainstem Klamath River Sites. Source: Asarian and Kann 2013.

For the period 1996 to 2007, average TN concentrations downstream from Iron Gate Dam (RM 193.1), vary by year, with mean concentrations of 1.2 mg/L (FERC 2007) and

a range of measured concentrations from less than 0.1 to over 2.0 mg/L (North Coast Regional Board 2010). Additional historical (1951 to 2001) nitrogen data is available as Total Kjeldahl Nitrogen or TKN, a measure of organic nitrogen plus ammonia. TKN median values for this period were 0.6 to 0.9 mg/L in the Lower Klamath Basin between Iron Gate Dam and Seiad Valley, with the highest median values occurring just downstream from the dam (PacifiCorp 2004b). Variability over the long-term record in this reach is relatively low compared with that of upstream reaches. For 1951 to 2001, high variability in nitrate concentrations is apparent in the reach between Iron Gate Dam and the Salmon River confluence, with some relatively high concentrations (greater than 5 mg/L) occurring at the Seiad Valley location (RM 132.7) (PacifiCorp 2004b).

More recent data from 2001 to 2015 show TN concentrations tend to decrease with distance downstream from Iron Gate Dam (Figures C-21, C-24, and C-26) and the nitrogen dynamics immediately downstream of Iron Gate Dam are affected by conditions within the Lower Klamath Project reservoirs (Asarian et al. 2010; Asarian and Kann 2013; Oliver et al. 2014; Watercourse Engineering, Inc. 2011a, 2011b, 2012, 2013, 2014, 2015, 2016). Based on data collected from 2001-2011, TN concentrations in the river downstream from Iron Gate Dam are generally lower than those in upstream reaches (Figures C-24 and C-26) due to dilution from the springs downstream from J.C. Boyle Dam and reservoir retention in the Hydroelectric Reach (Asarian et al. 2009; Asarian and Kann 2013). Further decreases in TN occur in the mainstem river downstream from Iron Gate Dam due to a combination of tributary dilution and in-river nitrogen removal processes such as denitrification and/or storage related to biomass uptake (Asarian et al. 2010). TN concentrations between 2001 and 2011 are highest in the low flow year 2001 and lowest in the high flow year 2011 (Asarian and Kann 2013). On a seasonal basis, TN increases from May through November, with peak concentrations (1−1.5 mg/L) typically observed during September and October (Figure C-24). Analysis of the 2001−2004 dataset also indicates that median TN concentrations in the Klamath River from Iron Gate Dam to (RM 193.1) to Seiad Valley (RM 132.7) exceed 0.2 mg/L (Asarian and Kann 2006b). A review of median TN concentrations from 2012 to 2015 (Figure C-21) also shows TN consistently exceeding 0.2 mg/L downstream of Iron Gate Dam (RM 193.1), Seiad Valley (RM 132.7), and Happy Camp (RM 108.4) (Watercourse Engineering, Inc. 2013, 2014, 2015, 2016).

Ratios of TN to TP (TN:TP) measured in the Klamath River suggest the potential for the system to be nitrogen-limited with some periods of co-limitation by N and P; however, concentrations of both nutrients are high enough in the river from Iron Gate Dam (RM 193.1) to approximately Seiad Valley (RM 132.7) (and potentially farther downstream) that nutrients are not likely to be limiting primary productivity (e.g., periphyton growth) in this portion of the Klamath River (FERC 2007, HVTEPA 2008, Asarian et al. 2010). In addition, nitrogen-fixing species dominate the periphyton communities in the lower reaches of the Klamath River where inorganic nitrogen concentrations are low (Asarian et al. 2010, Asarian et al. 2015). Since these species can fix their own nitrogen from the atmosphere, nitrogen would not limit their growth (Asarian et al. 2015).

Data collected during 2001 to 2015 indicate nitrate concentrations also tend to decrease longitudinally in the Klamath River downstream from Iron Gate Dam (Asarian et al. 2010; Asarian and Kann 2013; Oliver et al. 2014; Watercourse Engineering, Inc. 2011a, 2011b, 2012, 2013, 2014, 2015, 2016). Although patterns in nutrient concentrations vary between years, nitrate typically increases between July and September, with measured

concentrations downstream from Iron Gate Dam frequently greater than 0.25 mg/L (Asarian et al. 2010; Asarian and Kann 2013). In the fall, nitrate concentrations tend to increase again, occasionally reaching values of over 0.6 mg/L (Asarian et al. 2010, Asarian and Kann 2013). Mean 2000 to 2004 nitrate concentrations downstream from Iron Gate Dam were 0.15 to 0.44 mg/L between March and November, with the highest concentrations observed in early September (FERC 2007). Over the same time period, mean nitrate concentrations farther downstream near the confluence of the Shasta River (RM 179.5) had decreased to 0.02 to 0.36 mg/L, with peaks observed in early November (FERC 2007). Nitrate generally comprises less than 40 percent of the TN concentration throughout the lower Klamath River (Asarian et al. 2010). Nitrate concentrations at Seiad Valley (RM 132.7) from 2001 to 2011 consistently increase in the fall between August and November frequently exceeding 0.2 mg/L (Asarian and Kann 2013, Oliver et al. 2014).

As a result of the seasonal production of ammonia in anoxic hypolimnetic waters of the Lower Klamath Project reservoirs (Section C.3.1.1) and the high pH levels (greater than 7.5 pH units) measured seasonally downstream from Iron Gate Dam (YTEP 2005, North Coast Regional Board 2011), the North Coast Regional Board evaluated all available sampling data records as part of Klamath River TMDL development. The North Coast Regional Board analysis showed that for sampling events in which all three parameters (pH, ammonia, and water temperature) were collected simultaneously, no acute or chronic toxicity exceedances of the *Water Quality Control Plan for the North Coast Region* (Basin Plan) criteria for ammonia were indicated (North Coast Regional Board 2010). For the May to November sampling period in 2005 to 2008, ammonia concentrations in the Klamath River downstream from Iron Gate Dam were generally less than 0.3 mg/L and constituted less than 10 percent of the TN concentration (Asarian et al. 2010). Highest concentrations were measured during fall months downstream from Iron Gate Dam (RM 193.1), with late-fall ammonia concentrations generally increasing at this location and values increasing to above 0.2 mg/L during November 2006. For the period 2000 to 2004, mean ammonia levels of 0.13 mg/L were reported in Iron Gate Dam outflow (FERC 2007).

Although tributary dilution generally has a proportionally greater effect on nutrient concentration reductions in the Klamath River downstream from Iron Gate Dam, nutrient retention is an important component of overall nutrient dynamics in this reach (Asarian et al. 2010, Oliver et al. 2014). In a study of the June to October and July to September periods during 2005 to 2008, nutrient retention in the reach from Iron Gate Dam to the Klamath River Estuary was calculated after accounting for tributary dilution (Asarian et al. 2010). For the study, positive retention values represented seasonal removal of nutrients from the water column through storage in phytoplankton/plant biomass or denitrification, and negative retention represented an internal source of nutrients from sediment release or phytoplankton regeneration and nitrogen fixation. Retention rates downstream from Iron Gate Dam were variable but generally positive for TP, although negative retention was observed during some years in the reach between Seiad Valley (RM 132.7) and the Salmon River (RM 66.3), as well as further downstream to Turwar (RM 5.6) (Asarian et al. 2010). In general, TP and orthophosphate retention increased with distance downstream from Iron Gate Dam while particulate phosphorus retention decreased (i.e., negative retention). Nutrient retention for TN was similarly positive, with instances of negative retention observed during 2005 between Iron Gate Dam (RM 193.1) and Seiad Valley (RM 132.7) (see Section C.3.2.2 for discussion of retention in lower reaches). Additionally, during 2005 to 2008, total inorganic nitrogen (TIN = nitrite

+ nitrate + ammonia) retention was consistently positive between Iron Gate Dam and as far downstream as the Trinity River confluence (RM 43.3). The Asarian et al. (2010) analysis indicates that large quantities of nitrogen and phosphorus were retained in the river across the roughly 130 miles from Iron Gate Dam to just downstream from the Salmon River at Orleans (RM 58.9). During July to September of 2007 to 2008, the incoming nutrient load at Iron Gate Dam was reduced by 24 percent for TP, 25 percent for orthophosphate, 21 percent for particulate phosphorus, 41 percent for TN, 93 percent for TIN, and 21 percent for organic nitrogen (Asarian et al. 2010). Oliver et al. (2014) report a pattern of decreasing TN loads downstream of Iron Gate Dam due to algal sedimentation or denitrification during August to November, increasing TN loading during December through April once discharge increased in the winter/spring period, and relatively similar TN loads across sites from downstream of Iron Gate Dam to Seiad Valley between May and July. Trends in TP loads indicate TP is less influenced by algal blooms than seasonal changes in discharge with TP loads generally increasing with distance downstream of Iron Gate Dam at higher winter/spring discharge rates (Oliver et al. 2014).

C.3.2.2   Salmon River to Estuary

Downstream from the confluence with the Salmon River (RM 66.3), nutrient concentrations continue to decrease in the Klamath River as compared with those measured farther upstream. Historical (1950 to 2001) TP data indicate median values of 0.06 to 0.07 mg/L in river between the Salmon River confluence and near the Klamath River Estuary, with generally low variability (Figure C-25). Orthophosphate levels and variability over the long-term record in the reach downstream from the Salmon River are similar to those in the reach downstream from Iron Gate Dam (see previous section). Data from 2001 to 2015 indicate that TP concentrations in this reach are generally 0.05 to 0.1 mg/L with TP concentrations approaching 0.2 mg/L or greater in some years at the furthest upstream site Orleans (RM 58.9) (Figures C-22, C-23, and C-14; Asarian et al. 2010; Asarian and Kann 2013; Oliver et al. 2014; Watercourse Engineering, Inc. 2011a, 2011b, 2012, 2013, 2014, 2015, 2016). Data from 2005-2008 shows peak values occurring in September and October (Figure C-23). Water quality monitoring by the HVTEPA from 2008 to 2012 at Saints Rest Bar (RM 44.9) in the Klamath River shows TP ranging from 0.026-0.127 mg/L while SRP ranges from approximately 0.005 to 0.07 mg/L. Both TP and SRP (i.e., orthophosphate) at Saints Rest Bar generally increase from June through October reaching the annual peak between August and October (HVTEPA 2013). Data from 2007 to 2014 collected by the Yurok Tribe Environmental Program provide more recent seasonal patterns in the TP and SRP concentrations from Weitchpec (RM 43.6) upstream of the confluence with the Trinity River to the Klamath River Estuary (Sinnott 2008, 2009a, 2009b, 2010b, 2011b, 2012b, Hanington and Torso 2013; Hanington and Stawasz 2014; Hanington and Cooper-Carouseli 2014). In 2014, both TP and SRP at sites downstream of the Trinity River are at a minimum around mid-May, increase until peaking in September/October, and decrease until December (Hanington and Cooper-Carouseli 2014). Downstream from the Trinity River, orthophosphate often accounts for less than 50 percent of TP, possibly due to dilution from the Trinity River (Asarian et al. 2010).

As with upstream reaches, historical (1951 to 2001) nitrogen data is available as TKN, nitrate, and ammonia. TKN median values downstream from the Salmon River for this period were 0.25 to 0.3 mg/L (PacifiCorp 2004b). Variability over the 1951 to 2001 record in this reach is dependent on sampling location, with the greatest variability for

the most downstream site at Klamath Glen (RM 5.9). In the 1951 to 2001 dataset, high variability in nitrate concentrations is apparent throughout this reach, with some relatively high concentrations (greater than 3 mg/L) occurring at Orleans (RM 58.9) and Klamath Glen (RM 5.9) (PacifiCorp 2004b). Data from 2001 to 2015 indicate that TN concentrations in this reach are generally between 0.1 to 0.5 mg/L with TN concentrations frequently peaking above 0.5 mg/L between August and October at sites upstream of the Trinity River (Sinnott 2008, 2009a, 2009b, 2010b, 2011b, 2012b; Asarian and Kann 2013; Hanington and Torso 2013; Hanington and Stawasz 2014; Hanington and Cooper-Carouseli 2014; Watercourse Engineering, Inc 2012, 2013, 2014, 2015, 2016). However, TN concentrations substantially exceeded the typical range in some years like 2001. Data from 2005 to 2008 show the seasonal variations with TN increasing from May through November and peak concentrations (approximately 0.5 mg/L) typically observed during September and October (Figure C-24), which are at or above the Hoopa Valley Tribe numeric criterion of 0.2 mg/L TN. TN concentrations from 2008 to 2012 at Saints Rest Bar (RM 44.9) range from less than 0.1 mg/L to approximately 1.0 mg/L while nitrate plus nitrite ranges from less than 0.05 mg/L to approximately 0.28 mg/L. TN at Saints Rest Bar generally increases from June through October reaching the annual peak between September and October, but nitrate plus nitrite varies less until September when it increases and peaks between September and October (HVTEPA 2013). Downstream from the Trinity River confluence (RM 43.3), TN concentrations are typically less than 0.5 mg/L (YTEP 2005; Sinnott 2008, 2009a, 2009b, 2010b, 2011b, 2012b; Asarian and Kann 2013; Hanington and Torso 2013; Hanington and Stawasz 2014; Hanington and Cooper-Carouseli 2014), with general increases from spring to fall months. For the 2005 to 2008 dataset, TN increases were observed between September and October at Orleans (RM 58.9), upstream of the confluence with the Trinity River (approximately RM 43.3), and at Turwar (RM 5.6) (Figure C-24; Asarian et al. 2010). Similar patterns in TN concentrations were measured between 2009 and 2014 at sites between Orleans and Turwar (Sinnott 2008, 2009a, 2009b, 2010b, 2011b, 2012b; Asarian and Kann 2013; Hanington and Torso 2013; Hanington and Stawasz 2014; Hanington and Cooper-Carouseli 2014).

Nutrient retention rates in the Klamath River, from approximately the Salmon River confluence to the Trinity River, are variable for the period 2005 to 2008, but generally positive for TN and TP. However, from the Trinity River to the Klamath River Estuary, TN and TP nutrient retention rates are generally negative (Asarian et al. 2010). For example, during 2005 to 2008, total inorganic nitrogen (TIN = nitrite + nitrate + ammonia) retention was consistently negative between the Trinity River confluence and Turwar (RM 5.6) (Asarian et al. 2010). The Asarian et al. (2010) analysis suggests that while nitrogen and phosphorus are largely being removed from the river upstream of the Trinity River confluence (RM 43.3) during the June to October and July to September study periods, downstream from the confluence, nutrients are being added. Since retention is a load-based estimate and is inherently tied to flows, it is possible for nutrient loads to increase even while nutrient concentrations in the water decrease (in this case, only slightly).

C.3.2.3   Klamath River Estuary

Nutrient concentrations in the Klamath River Estuary are highly variable spatially and temporally and are greatly influenced by season, river flow, tidal prism, and location of the estuary mouth. In general, nutrient concentrations in the Klamath River Estuary are lower than in the Klamath River just upstream of the Trinity River confluence (RM 43.6)

and comparable to the nearest river sampling station (RM 5.6) near Turwar (YTEP 2004, 2005; Sinnott 2008, 2009a, 2009b, 2010b, 2011b, 2012b; Hanington and Torso 2013; Hanington and Stawasz 2014; Hanington and Cooper-Carouseli 2014). One exception to that trend is ammonia which is consistently higher in the Klamath River Estuary than upstream locations with peaks occurring between August and December (YTEP 2004, 2005; Sinnott 2009a, 2009b, 2010b, 2011b, 2012b; Hanington and Torso 2013; Hanington and Stawasz 2014; Hanington and Cooper-Carouseli 2014). Inter-annual and seasonal variability are apparent in the contemporary data collected by the Yurok Tribe during 2006 to 2014 (Sinnott 2008, 2009a, 2009b, 2010b, 2011b, 2012b; Hanington and Torso 2013; Hanington and Stawasz 2014; Hanington and Cooper-Carouseli 2014). For example, measured concentrations of TP in the Klamath River Estuary are below 0.12 mg/L during the period June to October 2006 to 2014. Contemporary data (2006 to 2014) indicate that TP concentrations in the Klamath River Estuary generally range from approximately 0.02 to 0.08 mg/L with peak values generally occurring in September and October, although 2009 and 2010 data indicated that concentrations of TP can continue to increase into November and December, especially during elevated river flows (Sinnott 2008, 2009a, 2009b, 2010b, 2011b, 2012b; Hanington and Torso 2013; Hanington and Stawasz 2014; Hanington and Cooper-Carouseli 2014). During peak concentrations, values often exceed the Hoopa Valley Tribe's standard of 0.035 mg/L TP (HVTEPA 2008). During the same period, orthophosphate is consistently reported at less than 0.06 mg/L. Orthophosphate often accounts for more than 50 percent of TP from June through October.

Contemporary data (2006 to 2014) indicate that TN concentrations in the Klamath River Estuary were consistently below 0.7 mg/L, generally ranging from approximately 0.1 to 0.6 mg/L (Sinnott 2008, 2009a, 2009b, 2010b, 2011b, 2012b; Hanington and Torso 2013; Hanington and Stawasz 2014; Hanington and Cooper-Carouseli 2014). Concentrations increase from June to October with peak values occurring in September and October, although 2009, 2010, and 2011 data indicate that concentrations can continue to increase into November and December, especially during high river flows (Sinnott 2008, 2009a, 2009b, 2010b, 2011b, 2012b; Hanington and Torso 2013; Hanington and Stawasz 2014; Hanington and Cooper-Carouseli 2014). During peak concentrations, values often exceed the Hoopa Valley Tribe's standard of 0.2 mg/L TN (HVTEPA 2008). During June to October 2006 to 2014, measured values of nitrate plus nitrite in the Klamath River Estuary are near or below the reporting limit (0.01 mg/L) with concentrations ranging from 0.01 mg/L to 0.2 mg/L though nitrate plus nitrite concentrations are typically between 0.01 mg/L and 0.04 mg/L. Concentrations of nitrate plus nitrite in the Klamath River Estuary increase from June to October, with peak values during this period occurring in September and October. As with TN, recent data indicates that nitrate plus nitrite concentrations can continue to increase into November and December, especially during elevated river flows (Sinnott 2010b, 2011b). Measured values of ammonia in the Klamath River Estuary were low, with measurements consistently below 0.1 mg/L during the period June to October 2006 to 2014, generally ranging from 0.01 mg/L to approximately 0.04 mg/L, with peak values generally occurring in September. Many ammonia samples from the Klamath River Estuary return values near or below the reporting limit of 0.01 mg/L (YTEP 2004, 2005; Sinnott 2008, 2009a, 2009b, 2010b, 2011b, 2012b; Hanington and Torso 2013; Hanington and Stawasz 2014; Hanington and Cooper-Carouseli 2014). However, the Klamath River Estuary sampling site has more detectable concentrations of ammonia than any other sampling site within the Yurok Reservation. Nutrient retention has not been explicitly measured in the Klamath River Estuary, although measurements have been made just

upstream of the Klamath River Estuary in the reach from the Trinity River confluence (RM 43.3) to Turwar (RM 5.6).

C.4   Dissolved Oxygen

C.4.1   Upper Klamath Basin

C.4.1.1   Keno Impoundment/Lake Ewauna

In the downstream Keno Impoundment/Lake Ewauna, dissolved oxygen reaches very low levels (less than 1 to 2 mg/L) during July through October as algae transported from Upper Klamath Lake settle out of the water and decay (see Figure 3.4-9). Decomposition of algae transported from Upper Klamath Lake appears to be the primary driver of low oxygen in the Keno Impoundment/Lake Ewauna. Dissolved oxygen concentrations measured in 2005 from the downstream end of Lake Ewauna to Keno Dam ranged from 7 to 8 mg/L in the early spring, and by late July concentrations were less than 2 mg/L throughout the water column (Deas and Vaughn 2006). During this same period, dissolved oxygen concentrations in Link River inflow were 7 to 8 mg/L, but apparently had little effect on the dissolved oxygen concentrations in the Keno Impoundment. Continuous dissolved oxygen data collected by Reclamation at Klamath River upstream of Keno Dam (USGS gage no. 11509370) for the period January 2006 through December 2018 exhibit seasonally low dissolved oxygen concentrations (less than 1 mg/L to 5 mg/L) from July through October.

C.4.1.2   Hydroelectric Reach

Dissolved oxygen concentrations in the Hydroelectric Reach vary on a seasonal and daily basis (e.g., FISHPRO 2000; PacifiCorp 2004b, 2008a; FERC 2007; Raymond 2008, 2009, 2010; USFWS 2008; Kirk et al. 2010; Zedonis and Turner 2010; Asarian and Kann 2011; Watercourse Engineering, Inc. 2011a, 2011b, 2012, 2013, 2014, 2015). Historical (1950 to 2001) data collected during daytime at various times in the year show median dissolved oxygen concentrations of approximately 5 to 8 mg/L in the Upper Klamath River between J.C. Boyle Reservoir and Iron Gate Dam, with the lowest median values occurring at RM 201.8 and RM 193.1 corresponding to Copco No. 1 Reservoir and Iron Gate Reservoir, respectively (Figure C-27). The highest historical median dissolved oxygen concentrations in the Hydroelectric Reach occur from RM 229.8 to RM 201.8 downstream of J.C. Boyle and upstream of Copco No. 1 Reservoir. Historical water column oxygen demand data is also available for the Hydroelectric Reach from J.C. Boyle to Iron Gate reservoirs. The historical record indicates relatively low biological oxygen demand (BOD) in the Lower Klamath Project reservoirs (Figure C-28).

During summer months, the Lower Klamath Project reservoirs exhibit varying degrees of dissolved oxygen supersaturation in surface waters due to high rates of phytoplankton photosynthesis and hypolimnetic anoxia as dissolved oxygen is depleted in bottom waters during seasonal thermal stratification and microbial decomposition of dead phytoplankton. While reaeration in the reach between Keno Dam and J.C. Boyle Reservoir can increase dissolved oxygen concentrations in the river to near saturation levels, the high oxygen demand in water entering J.C. Boyle Reservoir and a vertical water temperature gradient driven by diurnal fluctuations in river water temperature that limit vertical mixing from July to mid-September can still reduce dissolved oxygen concentrations in the reservoir itself (Figure C-29; Raymond 2008, 2009, 2010).

Peaking operations from J.C. Boyle Dam seem to have a negligible effect on dissolved oxygen concentrations between J.C. Boyle and Copco No. 1 because the free-flowing river upstream of J.C. Boyle and the high velocity of peaking flows provides sufficient aeration (PacifiCorp 2004b). Dissolved oxygen concentrations in Copco No. 1 and Iron Gate reservoirs vary seasonally with thermal stratification that limits mixing of surface and bottom waters and results in decreasing dissolved oxygen concentrations with depth. Copco No. 1 and Iron Gate reservoirs thermally stratify beginning in April or May and do not mix again until October to December with Iron Gate achieving complete mixing approximately a month later than Copco No. 1 (FERC 2007; Asarian and Kann 2011). During the stratification time period, dissolved oxygen concentrations in Iron Gate and Copco No. 1 surface waters generally are at or above saturation likely due to photosynthesis by aquatic plants and phytoplankton growth concentrating near the reservoirs' surface (Asarian and Kann 2011). Dissolved oxygen concentrations in the hypolimnetic waters near the bottom of the reservoirs are much lower reaching minimum values near 0 mg/L by July (for example, see 2008 data shown in Figures C-30 and C-31). Low dissolved oxygen concentrations (less than 3 mg/L) extend further up in the water column and longer in the season in Iron Gate Reservoir than in Copco No. 1 Reservoir (Figure C-32; Asarian and Kann 2011).
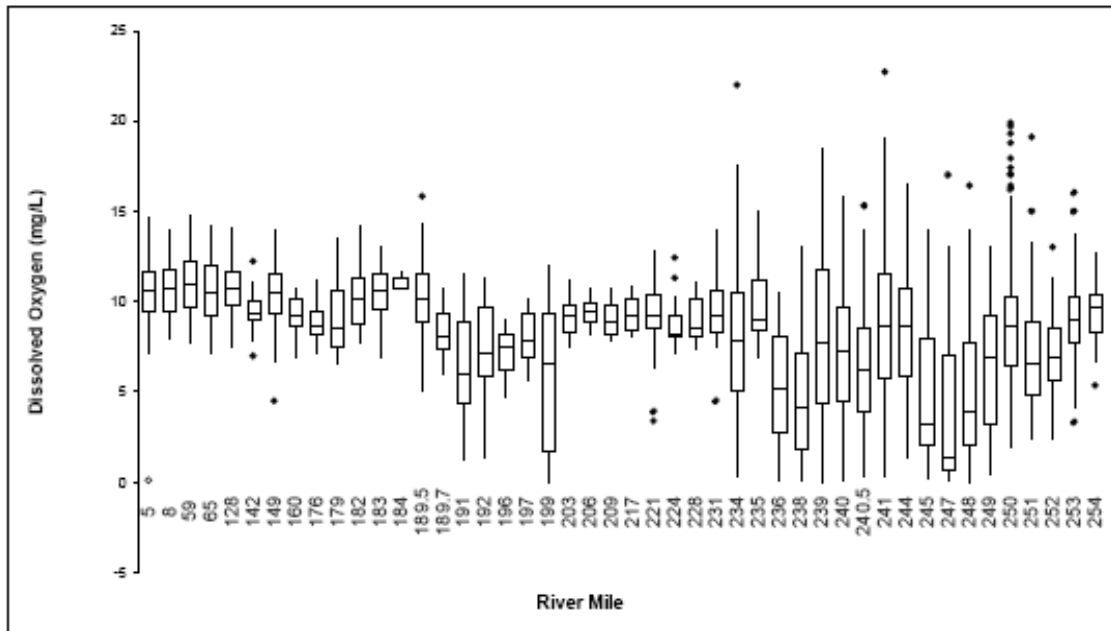
Figure C-27. Box and Whisker Plot of Historical Dissolved Oxygen Concentration Data Collected as Daytime Grab Samples from Various Sites in the Klamath River from Klamath River at Klamath Glen (RM 5.9) to Klamath River at Link River Dam (RM 259.7) between 1950 and 2001. Source: PacifiCorp 2004b.
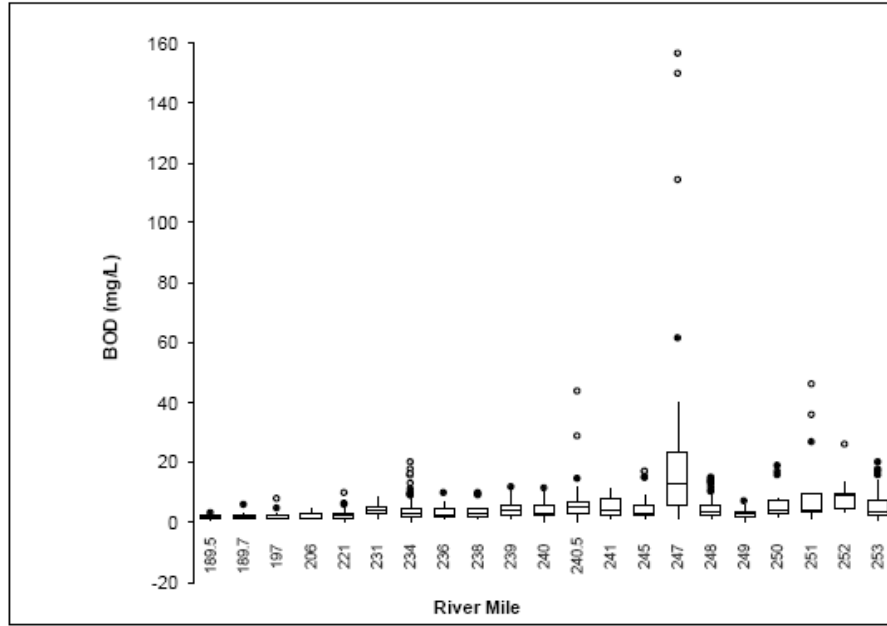
Figure C-28. Box and Whisker Plot of Historical BOD Data Collected from Riverine Sites in the Klamath River Between 1950 and 2001. J.C. Boyle Reservoir (RM 229.8 to RM 233.3), Copco No. 1 Reservoir (RM 201.8 to RM 208.3), Copco No. 2 Reservoir (RM 201.5 to RM 201.8), Iron Gate Reservoir (RM 193.1 to RM 200.0). Source: PacifiCorp 2004b.
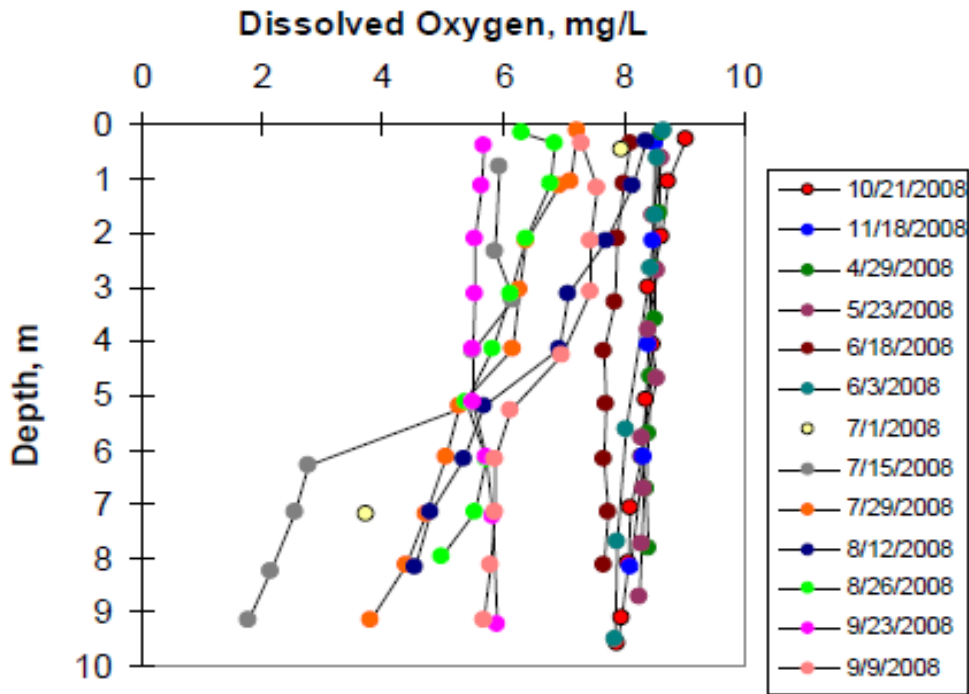
Figure C-29. Vertical Profiles of Dissolved Oxygen Concentration Measured in J.C. Boyle Reservoir Near the Dam in 2008. Source: Raymond 2009.
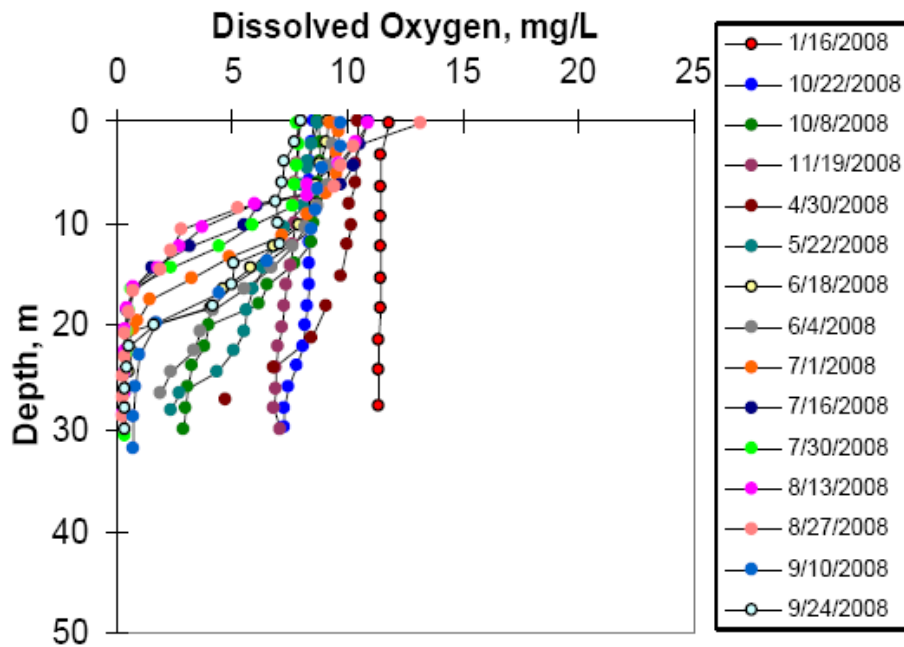
Figure C-30. Vertical Profiles of Dissolved Oxygen Concentration Measured in Copco No. 1 Reservoir Near the Dam in 2008. Source: Raymond 2009.
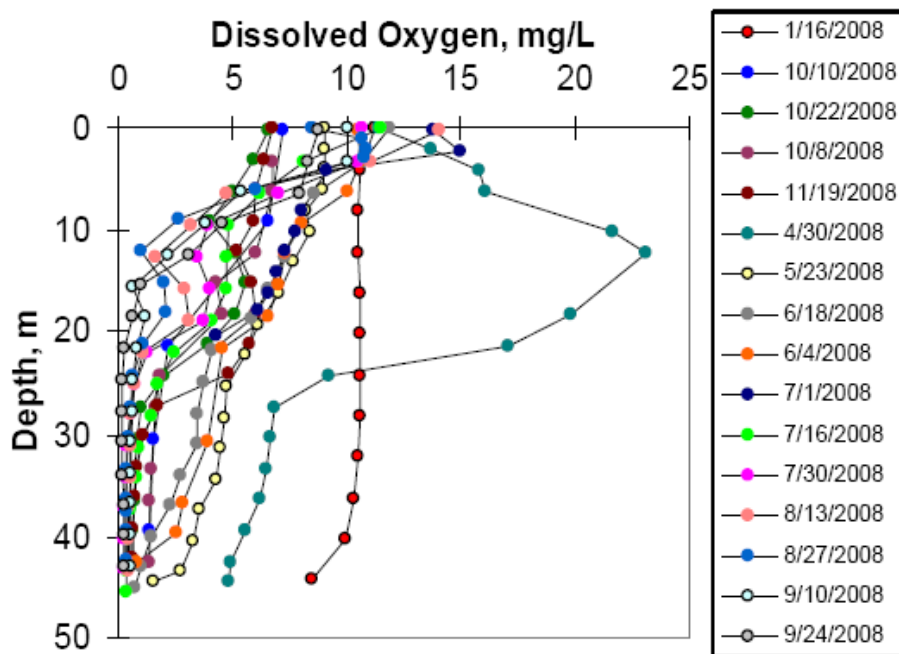
Figure C-31. Vertical Profiles of Dissolved Oxygen Concentration Measured in Iron Gate Reservoir Near the Dam (Bottom Plot) in 2008. Source: Raymond 2009.
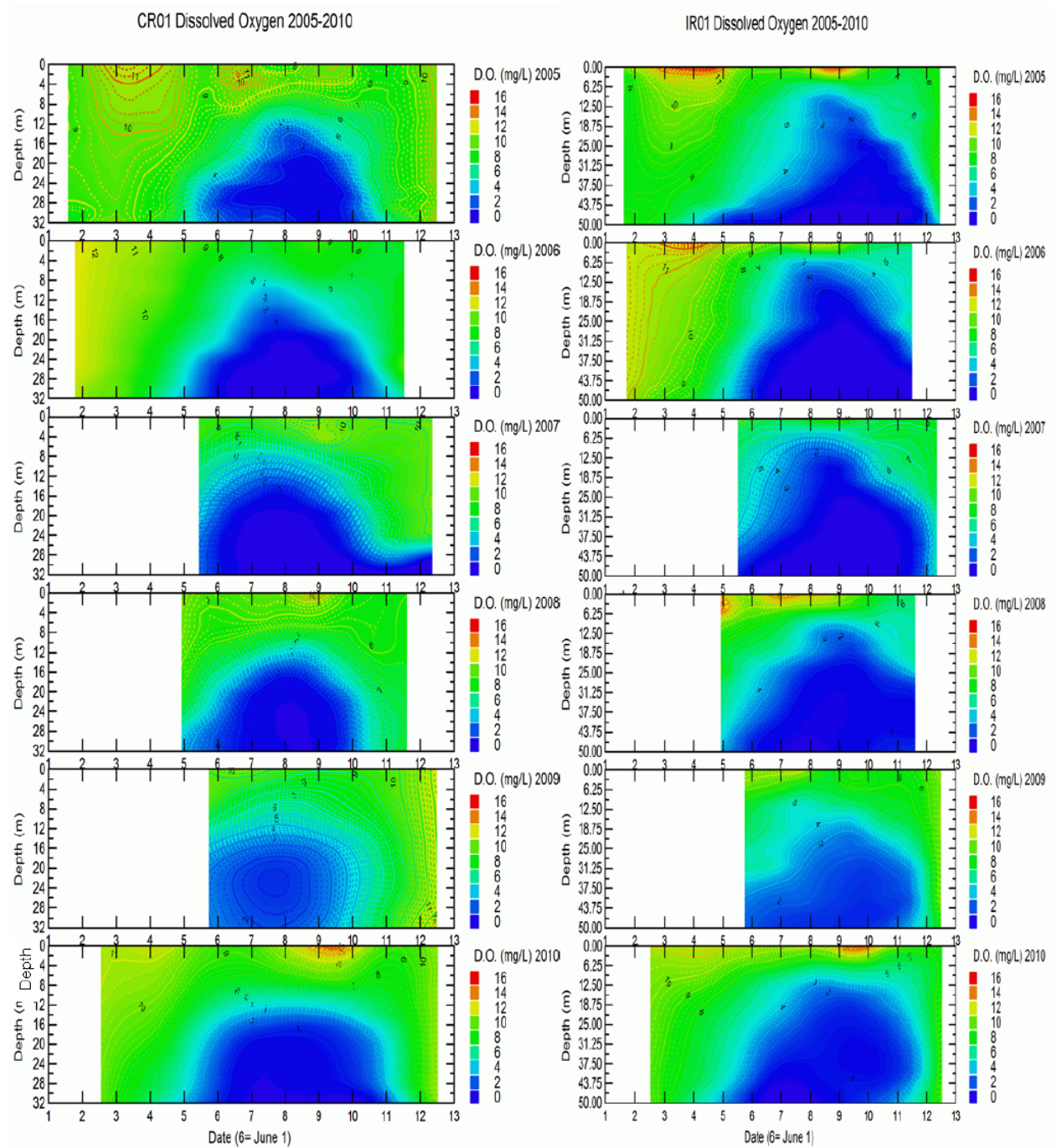
Figure C-32. Depth-time Distribution of Isopleths of Dissolved Oxygen Concentrations in Copco No. 1 Reservoir (CR01) and in Iron Gate Reservoir (IR01) from January 2005-December 2010. Source: Asarian and Kann 2011.

Substantial depression of dissolved oxygen concentrations with depth are reported in Iron Gate Reservoir as early as 1975 as part of the United States Environmental Protection Agency (USEPA) National Eutrophication Study (USEPA 1978). While

minimum dissolved oxygen concentrations generally occur coincident with maximum water temperatures in July and August, low dissolved oxygen concentrations (62 percent less than 8.0 mg/L and 11 percent less than 6.0 mg/L) and dissolved oxygen percent saturation (74 percent less than 90 percent saturation, 61 percent less than 85 percent saturation) occur in October at Iron Gate Reservoir. The lower dissolved oxygen concentrations and their occurrence later in the season at Iron Gate Reservoir is likely associated with the decomposition of cyanobacteria [blue-green algae] biomass as seasonal blooms decline and the breakdown of thermal stratification in the reservoir (Asarian and Kann 2013).

Daily patterns in dissolved oxygen concentrations follow a 24-hour cycle with photosynthesis by aquatic plants and phytoplankton elevating dissolved oxygen concentrations during the day and respiration by those same organisms decreasing dissolved oxygen concentrations at night. The daily dissolved oxygen concentration range has a seasonal trend typically peaking between late July and early September. The magnitude of the 24-hour variations in dissolved oxygen concentration is muted in Iron Gate Reservoir compared to other sections of the river due to the reservoir thermal mass and the depth water is withdrawn from the reservoir (Asarian and Kann 2013).

In addition to the biological oxygen demand from aerobic organisms in the water column itself, there is also a sediment oxygen demand that also that influences dissolved oxygen concentrations in the water column. Sediment oxygen demand is the rate at which dissolved oxygen is removed from the water column by the decomposition of organic matter in streambed or lakebed sediments. In lakes, reservoirs, and rivers, sediment oxygen demand can affect the level of dissolved oxygen concentrations in the water column (Doyle and Lynch 2005). Sediment oxygen demand ranges from approximately 1.75 to 3.25 grams of oxygen per square meter per day (g O_2/m^2/day) in sediment cores from three locations in J.C. Boyle sampled in 2002. The sediment oxygen demand from Copco No. 1 and Iron Gate reservoirs ranges from approximately 1.0 to 2.0 g O_2/m^2/day (FERC 2007). A comparison of the total oxygen required to meet the sediment oxygen demand and the water column biological oxygen demand in the reservoirs indicates that the sediment oxygen demand in J.C. Boyle was much less than water column biological oxygen demand and thus the sediments are not the largest influence on dissolved oxygen concentrations in J.C. Boyle Reservoir. Conversely, the sediment oxygen demand was greater than the water column biological oxygen demand in Copco No. 1 and Iron Gate reservoirs, indicating that sediment oxygen demand has a greater influence on water column dissolved oxygen concentrations in the two larger reservoirs (PacifiCorp 2004b).
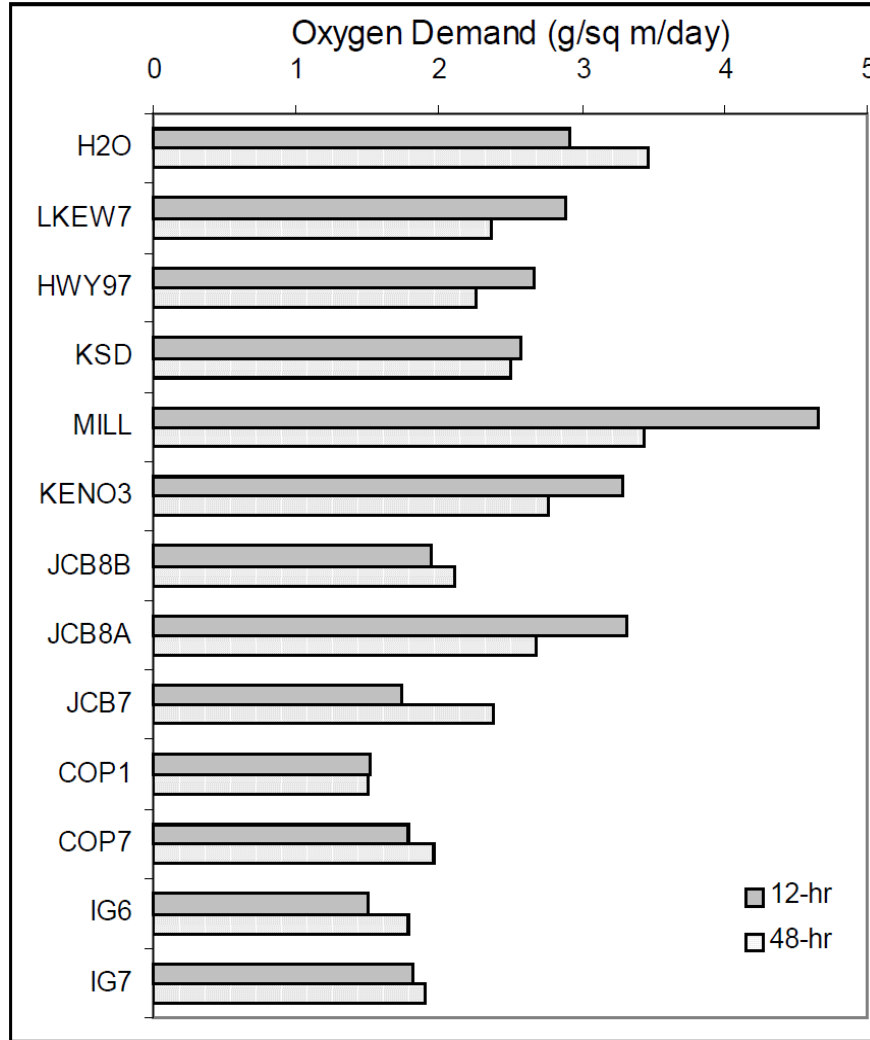
Figure C-33. Sediment Oxygen Demand from Sediment Cores at Multiple Locations within J.C. Boyle (JCB8B, JCB8A, JCB7), Copco No. 1 (COP1 and COP7), and Iron Gate (IG6 and IG7) Reservoirs. Source: PacifiCorp 2004b.

C.4.2   Mid- and Lower Klamath Basin

C.4.2.1   Iron Gate Dam to Salmon River

Historical (1950 to 2001) dissolved oxygen concentrations (reflecting day time grab sampling) in the Klamath River downstream of Iron Gate Dam are variable with dissolved oxygen concentrations approaching saturation values in Klamath River reaches that are free-flowing (PacifiCorp 2004b). Discharges occur from depths of approximately 12 meters in Iron Gate Reservoir, so downstream dissolved oxygen concentrations tend to reflect oxygen conditions of the reservoir's lower epilimnion (Section C.4.1.1) when the reservoir is stratified, with some increases in dissolved oxygen concentrations as the water is re-aerated upon discharge. In the fall, before and after reservoir turnover, low dissolved oxygen concentrations from the hypolimnion can

be translated downstream. Dissolved oxygen concentration median values between 1950 and 2001 were 8.1 to 10.8 mg/L in the Klamath River between Iron Gate Dam and the confluence with the Salmon River, with the lowest median values and the greatest general variability in the first mile downstream from the dam (Figure C-27, PacifiCorp 2004b).

More recent data indicates dissolved oxygen concentrations immediately downstream from Iron Gate Dam regularly fall below 8.0 mg/L and the current Basin Plan minimum dissolved oxygen criteria based on percent saturation (Figure C-34) (Karuk Tribe of California 2001, 2002, 2003, 2007, 2009, 2010a, 2010b, 2011, 2012, 2013; North Coast Regional Board 2010; Asarian and Kann 2011, 2013; Watercourse Engineering, Inc. 2011a, 2011b, 2012, 2013, 2014, 2015, 2016). Based on continuous sonde data collected at multiple locations in the lower Klamath River during summer 2004 to 2006, roughly 45 to 65 percent of measurements immediately downstream from Iron Gate Dam did not achieve the (previous) Basin Plan water quality objective of 8.0 mg/L (instantaneous minimum concentration). The Basin Plan water quality objective is now based on percent saturation. The percent of dissolved oxygen concentration measurements below 8.0 mg/L decreases with distance downstream, particularly in 2005 and 2006. Table C-2 summarizes the percent of dissolved oxygen concentrations in the Lower Klamath River below 8.0 mg/L during Summer 2004 to 2006.
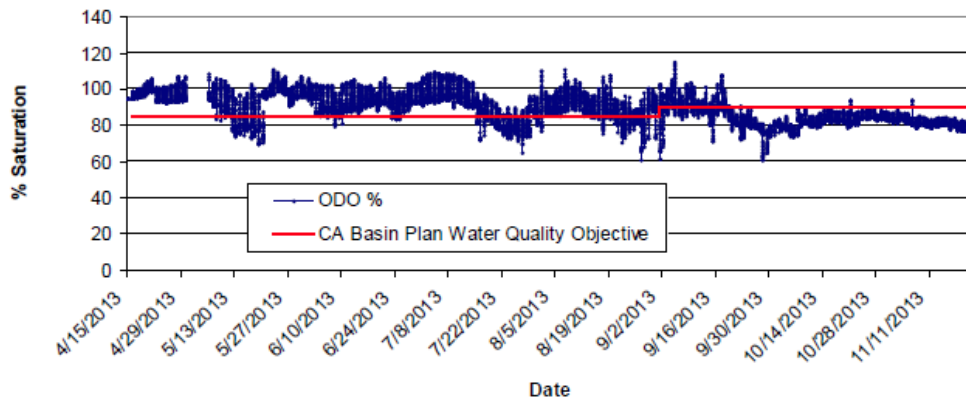
Figure C-34. Percent saturation dissolved oxygen readings recorded every 30-minutes for Klamath River below Iron Gate Dam (IG) in 2013. The red line indicates the Basin Plan Klamath River site-specific dissolved oxygen water quality objective from the Oregon-California state line to the mouth of the Scott River (greater than 90 percent saturation from October 1 to March 30 and greater than 85 percent saturation from April 1 to September 30. Source: Karuk Tribe of California (2013).

Continuous Sonde data collected between 2001 and 2017 show the range of daily average dissolved oxygen concentrations downstream from Iron Gate Dam (RM 193.1) between September through October and in November (Table C-3). Analysis of the longer dataset from 2001 to 2017 agrees with the 2004 to 2006 analysis results showing dissolved oxygen concentrations are less than 8.0 mg/L for June to October immediately downstream of Iron Gate Dam approximately 48 percent of the time. Asarian and Kann (2013) note that dissolved oxygen concentrations immediately downstream of Iron Gate Dam exceed water quality thresholds later in the year than other sites downstream of

Iron Gate Dam. *In situ* continuous data collected during 2008 to 2017 by PacifiCorp in the Klamath River downstream from Iron Gate Dam also demonstrate the seasonal decreases in dissolved oxygen (measured as percent saturation and concentration) originating from the reservoirs, with the lowest average monthly values occurring in August (warmer water temperatures) through November (cooler water temperatures), depending on the year (Table C-4).

In 2008, PacifiCorp began implementation of turbine venting at Iron Gate Dam as KHSA Interim Measure 3, with the goal of improving dissolved oxygen concentrations immediately downstream from the dam during periods of reservoir stratification. Early testing results in 2008 indicated that dissolved oxygen concentrations immediately downstream from Iron Gate Dam could be increased by approximately 0.5 to 2 mg/L (approximately 7 to 20 percent dissolved oxygen saturation) through the mechanical introduction of oxygen as water passed through the turbines (Carlson and Foster 2008, PacifiCorp 2008a). In 2009, PacifiCorp installed a forced air blower to enhance aeration, which became fully operational in 2010. The combination of turbine venting and the blower during the 2010 testing period increased dissolved oxygen immediately downstream of the dam from approximately 50 percent saturation to approximately 70 percent saturation. Further downstream, 2.5 river miles from the dam, the combination of turbine venting and the blower increased dissolved oxygen from approximately 65 percent saturation to approximately 80 percent saturation. By approximately six river miles downstream from the dam, natural river aeration increased dissolved oxygen concentrations sufficiently that there was no difference between periods with turbine venting plus blower operation and periods without (PacifiCorp 2011). Currently, and on a year-round basis, PacifiCorp automatically operates the blower when dissolved oxygen levels drop below 87 percent saturation and the blower is automatically turned off when dissolved oxygen levels exceed 87 percent saturation.

Despite the improvements reported during 2008 and 2011, and as discussed in the previous paragraphs, dissolved oxygen immediately downstream of Iron Gate Dam has continued to exhibit percent saturation values below the Basin Plan Basin Plan minimum dissolved oxygen criteria of 85 percent saturation for the period April 1 through September 30, and below the minimum criterion of 90 percent saturation for the period October 1 to March 31, with the majority of measured low dissolved oxygen saturation values occurring from August through November (PacifiCorp 2013, 2014, 2014, 2015, 2016, 2017, Karuk Tribe of California 2012, 2013).

Table C-2. Percent of Dissolved Oxygen Concentrations below 8.0 mg/L in the Lower Klamath River during Summer 2004 to 2006.

Location	2004		2005		2006	
	n [1]	%	n [1]	%	n [1]	%
At Iron Gate Dam (RM 193.1)	2,706	64	4,498	45	5,391	61
Upstream of Shasta River (RM 179.5)	5,478	50	5,533	49	-	-
Upstream of Scott River (RM 145.1)	2,966	58	4,457	47	-	-
Seiad Valley (RM 132.7)	3,381	57	4,713	45	5,526	40
Orleans (RM 58.9)	57	37	4,533	23	5,349	15
Weitchpec (RM 43.6)	4,142	48	5,400	7	5,332	6
Downstream from Weitchpec (≈ RM 43.4)	5,500	16	3,529	11	5,293	4
Upstream of Trinity (RM 43.3)	-	-	5,535	5	5,739	3
Turwar (RM 5.6)	5,066	30	5,543	6	-	-

Source: Ward and Armstrong 2006, North Coast Regional Board 2010.
[1] Dissolved oxygen measurements were collected at 30-minute increments for a total of forty-eight daily measurements.
Key:
    n=number of measurements
    %=percent of measurements not achieving the Basin Plan previous water quality objective of 8.0 mg/L

Table C-3. Range of Observed Dissolved Oxygen Concentrations Downstream from Iron Gate Reservoir.

Year	General Range of Daily Average Dissolved Oxygen (mg/L) downstream from Iron Gate Dam (RM 193.1, near USGS Gage No. 11516530)		
	September–October	November	Source
2001	4–6[1]	7–8	Karuk Tribe of California 2003
2002	4–9[2]	-	Karuk Tribe of California 2003
2004	6–9.5	8–9	Zedonis and Turner 2010
2006	6.5–8	7–8	Karuk Tribe of California 2009
2007	7–9[2]	-	Karuk Tribe of California 2009
2008	6.5–8.5	-	Karuk Tribe of California 2009
2009	7.5–10	-	Karuk Tribe of California 2010a
2010	7–9.5	-	Karuk Tribe of California 2010b
2011	7–9.5	8–9.5	Karuk Tribe of California 2011
2012	6–9.5	7.5–8.5	Karuk Tribe of California 2012
2013	7–9	8.5–9.5	Karuk Tribe of California 2013
2014	6–13	8.5–11	PacifiCorp 2014
2015	7–9	7–9	PacifiCorp 2015
2016	6–9.5	8–9	PacifiCorp 2016
2017	5–10	8–9.5	PacifiCorp 2017

[1] No September data reported
[2] No October data reported
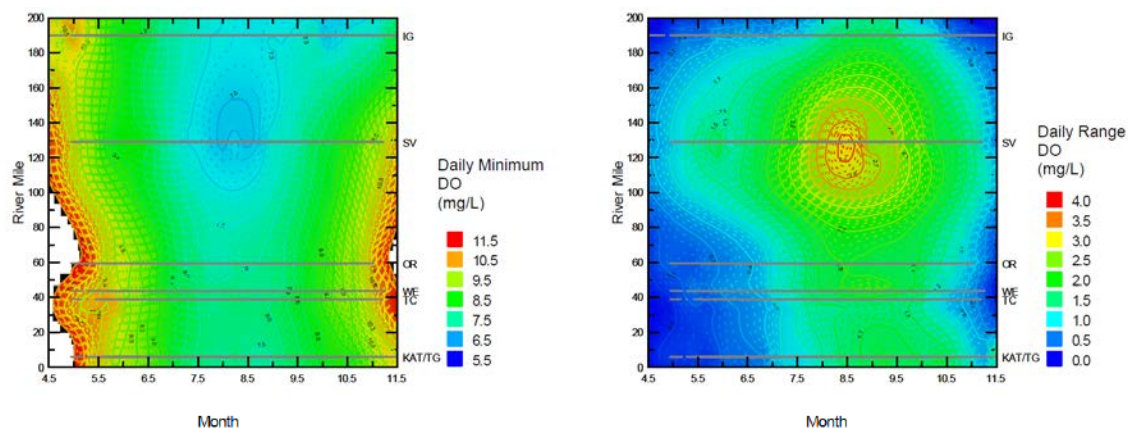
Figure C-35. Distribution of Isopleths Showing the 2001-2011 Average Daily Minimum Dissolved Oxygen (DO) Concentrations and Daily Range of Dissolved Oxygen (DO) Concentrations at Klamath River Sites Downstream of Iron Gate Dam from Around Mid-April (4.5) to Mid-November (11.5). Grey Lines Represent the Times Dissolved Oxygen Was Measured in the Klamath River Downstream of Iron Gate Dam (IG at RM 193.1), at Seiad Valley (SV at RM 132.7), at Orleans (OR at RM 58.9), at Weitchpec Upstream of the Trinity River (WE at RM 43.6), Upstream of Tully Creek (TC at RM 40.1), and at Turwar (KAT/TG at RM 5.6). Source: Asarian and Kann 2013.

It has been suggested that daily fluctuations of up to 3 mg/L measured in the Klamath River downstream from Iron Gate Dam (RM 193.1) (Karuk Tribe of California 2002, 2003; YTEP 2005; North Coast Regional Board 2010; Asarian and Kann 2011, 2013) are caused by daytime phytoplankton photosynthesis and nighttime respiration. Low dissolved oxygen concentrations can also be often caused by bacterial aerobic decomposition of phytoplankton in the reservoir (Ward and Armstrong 2010).

Table C-4. Average Monthly Water Temperature, Dissolved Oxygen Percent Saturation, and Dissolved Oxygen Concentration in the Klamath River Downstream from Iron Gate Dam (RM 193.1).

Month	Average Monthly Water Temperature (ºC)	Average Monthly Dissolved Oxygen (% Saturation)	Average Monthly Dissolved Oxygen (mg/L)
2008			
June	18.4	92.2	8.7
July	22.3	90.0	7.8
August	21.8	91.8	8.0
September	18.6	84.5	7.9
October	14.8	66.2	6.7
November	10.3	67.4	7.5
December	7.0	70.0	8.5
2009			
January	3.7	79.4	10.5
February	4.4	83.0	10.8
March	6.7	83.2	10.2
April	8.4	82.2	9.6
May	17.4	94.4	9.0
June	19.3	87.9	8.1

Month	Average Monthly Water Temperature (ºC)	Average Monthly Dissolved Oxygen (% Saturation)	Average Monthly Dissolved Oxygen (mg/L)
July	21.2	86.8	7.7
August	21.7	99.9	8.8
September	19.4	95.7	8.8
October	14.6	77.7	7.9
November	9.9	71.2	8.1
December	5.0	81.2	10.4
2010			
January	3.9	94.2	11.4
February	5.4	100.2	11.1
March	7.2	96.6	10.5
April	9.5	108.9	11.4
May	12.7	104.8	10.2
June	16.8	94.9	8.5
July	21.3	98.8	8.1
August	21.9	95.9	7.7
September	18.4	105.2	9.1
October	15.5	92.5	8.5
November	11.8	62.5	6.2
2011			
January	4.0	90.2	10.9
February	4.6	95.0	11.3
March	5.8	96.3	11.1
April	9.2	101.4	10.7
May	12.9	107.1	10.4
June	16.1	100.9	9.2
July	20.6	94.0	7.8
August	22.2	97.9	7.8
September	20.2	93.4	7.8
October	16.0	92.1	8.4
November	10.1	74.6	7.8
December	5.6	90.2	10.5
2012			
January	3.8	12.0	90.8
February	4.3	12.6	96.2
March	5.8	12.0	95.7
April	9.4	10.6	92.2
May	15.5	9.5	94.9
June	17.8	8.8	92.2
July	20.9	8.2	91.7
August	21.9	7.5	85.0
September	19.5	7.7	83.1
October	16.1	7.2	72.9
November	11.3	8.0	72.3
December	7.2	9.7	79.8
2013			
January	15.2	97.8	9.1
February	8.5	96.6	10.4
March	11.4	102.4	10.3
April	16.0	107.8	9.8
May	18.9	105.6	9.0
June	21.7	101.6	8.2

Month	Average Monthly Water Temperature (ºC)	Average Monthly Dissolved Oxygen (% Saturation)	Average Monthly Dissolved Oxygen (mg/L)
July	21.4	100.3	8.2
August	19.9	92.5	7.8
September	14.2	89.5	8.5
October	10.0	89.6	9.4
November	5.3	91.2	10.7
December	15.2	97.8	9.1
2014			
January	3.8	94.0	11.5
February	4.7	99.3	11.9
March	8.4	98.6	10.9
April	12.0	97.1	9.9
May	16.7	101.4	9.3
June	19.3	101.0	8.7
July	22.3	103.6	8.6
August	22.0	111.1	9.8
September	19.3	104.7	9.7
October	15.8	80.3	8.0
November	11.0	87.1	9.7
December	7.6	91.0	10.9
2015			
January	5.7	93.5	11.7
February	-	-	-
March	10.3	103.5	11.6
April	12.0	93.4	10.1
May	15.9	96.9	9.6
June	20.0	94.4	8.6
July	22.4	91.9	8.0
August	20.9	83.3	7.4
September	18.7	80.0	7.5
October	15.8	79.5	7.9
November	11.6	78.2	8.5
December	6.8	85.3	10.4
2016			
January	4.3	103.8	12.5
February	5.2	101.3	11.9
March	7.7	103.6	11.4
April	12.4	101.7	10.1
May	15.7	105.9	9.7
June	19.0	103.3	8.9
July	20.2	93.7	7.9
August	21.4	81.9	6.7
September	18.7	77.8	7.2
October	15.0	84.1	8.1
November	11.3	87.1	8.9
December	7.0	91.3	10.3
2017			
January	2.9	90.3	12.2
February	4.3	97.8	12.7
March	7.2	102.7	12.4
April	10.2	106.1	11.2
May	14.7	94.2	9.6

Month	Average Monthly Water Temperature (°C)	Average Monthly Dissolved Oxygen (% Saturation)	Average Monthly Dissolved Oxygen (mg/L)
June	19.4	97.5	8.5
July	21.9	89.3	7.8
August	22.3	79.2	6.9
September	19.9	74.0	6.7
October	14.3	73.4	7.5
November	10.5	78.8	8.8
December	6.6	84.6	10.4

Raw daily data from http://www.pacificorp.com/es/hydro/hl/kr.html# (PacifiCorp 2008b, 2009, 2010b, 2011, 2012, 2013, 2014, 2015 2016, 2017). Data obtained with YSI 6600 V2 or 6900 Multiprobe DataSondes (30-minute intervals).

Farther downstream in the mainstem Klamath River, near Seiad Valley (RM 132.7), dissolved oxygen concentrations tend to increase; however, values below 8.0 mg/L do still occur (i.e., 2001, 2002, 2006, 2007, 2010, 2011, 2012, 2013, 2014, and 2015 as reported in Karuk Tribe of California [2001, 2002, 2009, 2010b, 2011, 2012, 2013] and Watercourse Engineering, Inc. [2011a, 2011b, 2012, 2013, 2014, 2015, 2016]). Dissolved oxygen concentrations near Seiad Valley continue to exhibit variability, with mean daily values ranging from approximately 6.5 mg/L to supersaturated concentrations of approximately 11.5 mg/L from June through November 2001 to 2002 and 2006 to 2013 (Karuk Tribe of California 2001, 2002, 2003, 2007, 2009, 2010a, 2010b, 2011, 2012, 2013). Longitudinal variations in dissolved oxygen concentrations from Iron Gate Dam (RM 193.1) to Seiad Valley (RM 132.7) are most pronounced between mid-September and November when dissolved oxygen concentrations decrease immediately downstream of Iron Gate but increase at Seiad Valley (Karuk Tribe of California 2013). Dissolved oxygen concentrations at Seiad Valley between 2001 and 2011 are less than the 8.0 mg/L, 90 percent saturation, and 85 percent saturation water quality thresholds less frequently than immediately downstream of Iron Gate Dam, but still are less than these water quality thresholds more frequently than further downstream sites (Asarian and Kann 2013). Between July and September, dissolved oxygen concentrations at Seiad Valley are less than 8.0 mg/L for 29 to 51 percent of measurements, less than 90 percent saturation for 28 to 34 percent of measurements, and less than 85 percent saturation for 10 to 18 percent of measurements (Asarian and Kann 2013). More contemporary data from 2012 to 2016 have dissolved oxygen concentration patterns similar to previous measurements with concentrations ranging from approximately 6.0 mg/L to (supersaturated concentrations of) approximately 12 mg/L from June through November and dissolved oxygen concentrations regularly less than 8.0 mg/L during that time period (Watercourse Engineering, Inc. 2013, 2014, 2015, 2016).

C.4.2.2   Salmon River to Estuary

Measured dissolved oxygen concentrations in the mainstem Klamath River downstream from the confluence with the Salmon River (RM 66.3) continue to increase relative to concentrations at upstream sites (Figure C-36). Despite this, values sometimes fall below 8.0 mg/L in this reach (e.g., at the Orleans gage [RM 58.9] during 2001, 2002, 2006, 2012, 2013 as reported in Karuk Tribe of California [2001, 2002, 2007, 2009, 2010a, 2010b, 2011, 2012, 2013], Ward and Armstrong 2006, North Coast Regional Board 2010). Dissolved oxygen concentrations near Orleans also exhibit variability, with

mean daily values ranging from approximately 6.5 mg/L to supersaturated concentrations of 11.5 mg/L from June through November in 2001, 2002, and 2006 to 2013. Asarian and Kann (2013) report dissolved oxygen concentrations at Orleans are less than 8.0 mg/L for 16 percent of measurements, less than 90 percent saturation for 10 percent of measurements, and less than 85 percent saturation for only 1 percent of measurements between June and October from 2001 to 2011. Dissolved oxygen concentrations were most frequently less than the 8.0 mg/L, 90 percent saturation, and 85 percent saturation water quality thresholds during July and August when water temperature peaked (Asarian and Kann 2013). Extremely high mean daily dissolved oxygen concentrations (11 to 15.5 mg/L) (Sonde data) were reported for October 2006 at the Orleans gage (Karuk Tribe of California 2007, 2009). More contemporary grab sample data from 2012 to 2015 agree with previous measurements showing dissolved oxygen concentrations generally increase from Seiad Valley (RM 132.7) to Orleans (RM 58.9) with dissolved oxygen concentrations ranging from approximately 7.5 mg/L to supersaturated concentrations of approximately 11.5 mg/L between June and November with the minimum concentrations occurring in July or August (Figure C-36; Watercourse Engineering, Inc. 2013, 2014, 2015, 2016).

Dissolved oxygen concentrations in the mainstem Klamath River upstream of the confluence with the Trinity River (RM 43.3) ranged from approximately 5.5 to 10.3 mg/L in 2004, with the lowest concentration of dissolved oxygen occurring in September and the highest dissolved oxygen concentration occurring in October. Dissolved oxygen concentrations below 8.0 mg/L (the Basin Plan minimum dissolved oxygen criterion prior to 2010) occurred for extended periods of time in mid-August (8/13 to 8/22) and early September (8/30 to 9/8) (YTEP 2005). In 2009 at this location, dissolved oxygen concentrations ranged from approximately 7.1 to 11.8 mg/L, with minimum dissolved oxygen concentrations dropping below 8.0 mg/L (the Basin Plan minimum dissolved oxygen criterion prior to 2010) for an extended period of time from mid-July to early August, and again from late August to early September (Sinnott 2010a). In 2010, dissolved oxygen concentrations ranged from 7.9 to 12.1 mg/L (Sinnott 2011a), with minimum dissolved oxygen concentrations remaining above the 2010 amended Basin Plan minimum dissolved oxygen concentration criteria based on percent saturation (e.g., 7.0, 6.9, and 7.8 mg/L for July, August, and September, respectively, see Section 3.2, Table 3.2-5). A synthesis of dissolved oxygen concentration data from 2001 to 2011 at Weitchpec (RM 43.6) on the mainstem Klamath River upstream of the confluence with the Trinity River indicates dissolved oxygen concentrations are below 8.0 mg/L most frequently between July and August, but the dissolved oxygen percent saturation is most frequently below 90 percent or 85 percent in August and September (Asarian and Kann 2013). On average, between June and October from 2001 to 2011, dissolved oxygen concentrations at Weitchpec (RM 43.6) are less than 8.0 mg/L for 16 percent of measurements, less than 90 percent saturation for 12 percent of measurements, and less than 85 percent saturation for only 5 percent of measurements. While only based on discrete dissolved oxygen concentration measurements during water quality grab samples, dissolved oxygen concentrations at Weitchpec (RM 43.6) from 2011 to 2015 are similar to previous measurements with concentrations ranging from approximately 8.0 mg/L to supersaturated concentrations of approximately 11.5 mg/L between June and November (Figure C-36; Sinnott 2010a, 2011a, 2012a; Watercourse Engineering, Inc. 2011a, 2011b, 2012, 2013, 2014, 2015; Hanington 2013; Hanington and Ellien 2013).
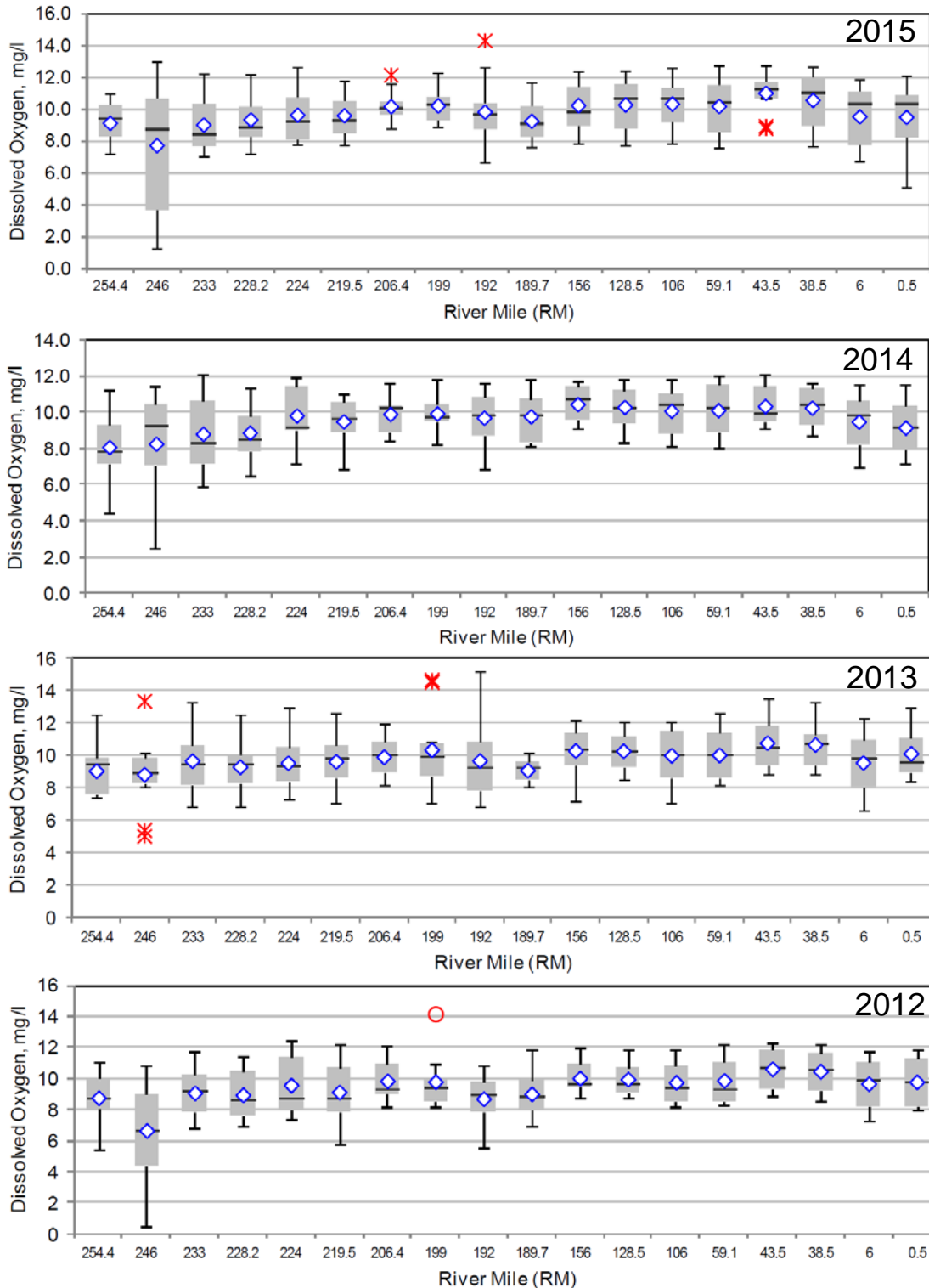
Figure C-36. Klamath River dissolved oxygen concentration trends from measurements made between February and December with median (-), mean (◊), outlier (*), and extreme outliers (○) identified. River miles specified are based on those accurate at the time of the reports and differ slightly from 2018 river mile designations (Table 3.2-1). Source: Watercourse Engineering, Inc. 2013, 2014, 2015, 2016.

Dissolved oxygen concentrations tend to decrease downstream from the confluence of the Trinity River with the Klamath River (Sinnott 2010a, 2011a, 2012a; Watercourse Engineering, Inc. 2011a, 2011b, 2012, 2013, 2014, 2015; Asarian and Kann 2013; Hanington 2013; Hanington and Ellien 2013). At the confluence with the Trinity River (RM 43.3) and at the Turwar gage (RM 5.6), daily minimum dissolved oxygen concentrations at the Trinity River and Turwar sites during May through November are consistently observed to occur late-night or early in the morning, likely due to respiration by aquatic vegetation (YTEP 2004, 2005; Sinnott 2010a, 2011a). At Turwar (RM 5.6) in 2004, minimum dissolved oxygen concentrations dropped below 8.0 mg/L (the Basin Plan minimum dissolved oxygen criterion prior to 2010) between late July and late August (YTEP 2005); dissolved oxygen concentrations ranging 5.9 to 10.1 mg/L were observed in August and September. In 2009, dissolved oxygen concentrations at Turwar ranged from 7.3 to 11.7 mg/L, with minimum dissolved oxygen concentrations dropping below 8.0 mg/L for an extended period of time from mid-July to early August (Sinnott 2010a). In 2010, dissolved oxygen concentrations ranged from 7.8 to 11.8 mg/L, with minimum values remaining above 2010 amended Basin Plan minimum dissolved oxygen concentration criteria based on percent saturation (e.g., 7.0, 6.9, and 7.8 mg/L for July, August, and September, respectively, see Section 3.2, Table 3.2-5). Dissolved oxygen concentration data from 2001 to 2011 downstream of the Trinity River (RM 43.3) and at Turwar (RM 5.6) show dissolved oxygen concentrations are the lowest and most frequently below the 8.0 mg/L, 90 percent saturation, and 85 percent saturation water quality thresholds during July and August (Asarian and Kann 2013). Turwar (RM 5.6) consistently has lower dissolved oxygen concentrations and is more frequently below the 90 percent or the 85 percent saturation levels than the site immediately downstream of the confluence with the Trinity River (RM 43.3) (Asarian and Kann 2013). Dissolved oxygen concentrations downstream of the confluence with the Trinity River (RM 43.3) to Turwar (RM 5.6) from 2011 to 2015 are similar to previous measurements in that reach. During that period, dissolved oxygen concentrations range from approximately 6.5 mg/L to supersaturated concentrations of approximately 11.5 mg/L between June and November with lower dissolved oxygen concentrations occurring more frequently at Turwar and higher dissolved oxygen concentrations occurring more frequently immediately downstream of the confluence with the Trinity River (Figure C-36; Sinnott 2010a, 2011a, 2012a; Watercourse Engineering, Inc. 2011a, 2011b, 2012, 2013, 2014, 2015; Hanington 2013; Hanington and Ellien 2013; Hanington and Cooper Carouseli 2014).

C.4.2.3   Klamath River Estuary

Dissolved oxygen concentrations within the Klamath River Estuary are highly variable spatially and temporally and are greatly influenced by season, river flow, vertical water column stratification (thermal and/or chemical), and location of the estuary mouth, the latter changing due to periodic sand bar movement. Dissolved oxygen concentrations have been monitored in the Klamath River Estuary by CDFG (Wallace 1998) and most recently by the Yurok Tribe Fisheries (Hiner 2006) and Environmental Programs (YTEP 2005) with support from the North Coast Regional Water Quality Control Board. Concentrations in the deeper, main channel of the estuary are generally greater than approximately 6.0 to 7.0 mg/L throughout the year (Hiner 2006; YTEP 2004, 2005).

Lower dissolved oxygen concentrations (ranging 2.5 to 5.5 mg/L) have been measured near the bottom of deep pools or in heavily vegetated side channels (Wallace 1998).

Low dissolved oxygen concentrations (less than 1.0 mg/L to approximately 5.0 mg/L) have been observed during summer months in the relatively shallow, heavily vegetated south slough (Wallace 1998, Hiner 2006). The low levels of dissolved oxygen observed in the slough are likely due to high rates of growth and subsequent decomposition of phytoplankton and aquatic plants, which are not abundant elsewhere in the Klamath River Estuary.

Dissolved oxygen concentrations become progressively more variable and generally lower nearer the estuary bottom and the estuary mouth, with concentrations frequently below 6.0 mg/L during summer months (YTEP 2004, 2005; Hiner 2006). Low dissolved oxygen has also been observed during late summer months when a sand berm forms across the river mouth, forcing the river to flow south diagonally between two sand spits. This berm prevents ocean water from entering the Klamath River Estuary, creating 'lagoon-like' conditions until higher flows breech the berm (Wallace 1998, Hiner 2006). These conditions were documented in 1994 and 2001; in 2001, a decrease in dissolved oxygen concentrations was measured related to sand berm formation, with especially marked decreases in dissolved oxygen concentrations in the south slough (Hiner 2006).

Additional monitoring by the Yurok Tribe Environmental Program from 2009 to 2015 detail more recent dissolved oxygen concentration trends near the surface in the lower Klamath River Estuary (RM 0.5). During that time period, dissolved oxygen concentrations usually range from approximately 7 mg/L to (supersaturated concentrations of) approximately 11 mg/L between June and November, with a particularly low value of 5.08 mg/L recorded on June 10, 2015 (Sinnott 2010a, 2011a, 2012a; Watercourse Engineering, Inc. 2011a, 2011b, 2012, 2013, 2014, 2015, 2016; Hanington 2013; Hanington and Ellien 2013; Hanington and Cooper Carouseli 2014). The dissolved oxygen percent saturation between 2009 and 2014 ranges from approximately 83 percent to 106 percent, with the percent saturation in 2014 exhibiting almost that entire range and broad seasonal variability (Figure C-37; Sinnott 2010a, 2011a, 2012a; Hanington 2013; Hanington and Ellien 2013; Hanington and Cooper Carouseli 2014).

Eilers and Raymond (2005) report sediment oxygen demand from a sediment core taken in the Klamath River Estuary in 2004. The raw sediment oxygen demand in the first 24-hours is 3.21 g O_2/m^2/day while the corrected sediment oxygen demand is 2.06 g O_2/m^2/day. The long-term rate of sediment oxygen demand is approximately 0.5 g O_2/m^2/day after 600 hours of incubation which is attributed to the high proportion of sand in the estuary sediment core (Eilers and Raymond 2005).
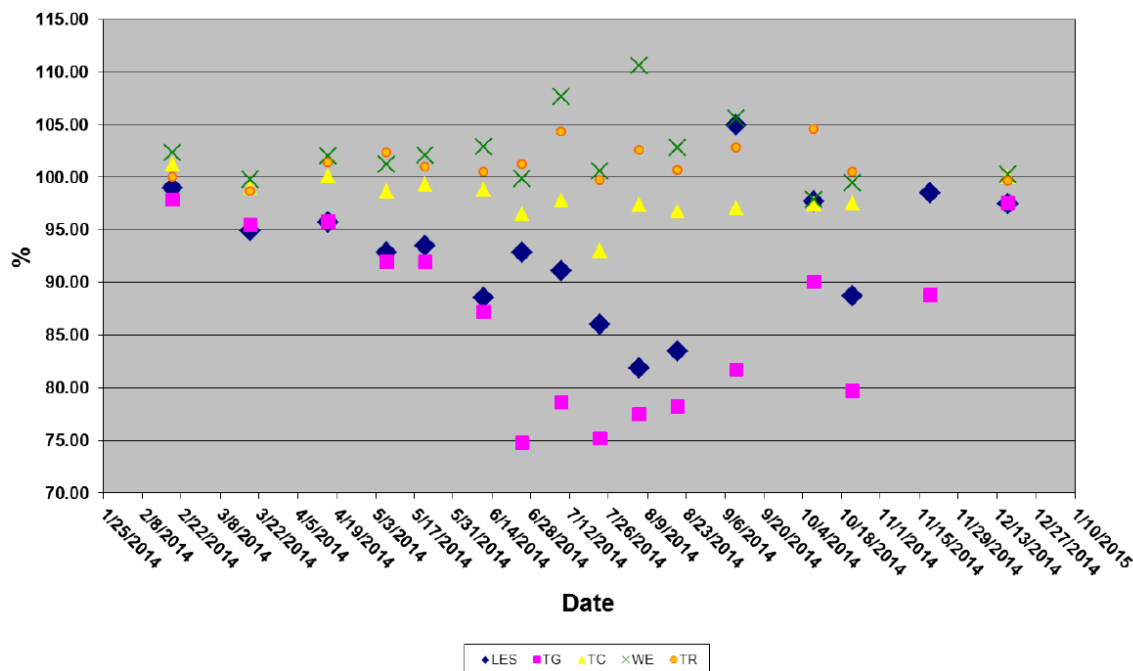
Figure C-37. Percent Saturation of Dissolved Oxygen During Grab Samples at the Lower Estuary Surface (LES [RM 0]), the Klamath River at Turwar Boat Ramp (TG [RM 6]), the Klamath River Upstream of Tully Creek (TC [RM 40.1]), the Klamath River at Weitchpec (Upstream of Trinity River) (WE [RM 43.6]), and the Trinity River Upstream the Confluence with the Klamath River (TR). Source: Hanington and Cooper-Carouseli 2014.

C.5   pH

C.5.1   Upper Klamath Basin

C.5.1.1   Hydroelectric Reach

Based upon monitoring conducted by PacifiCorp, pH in the Hydroelectric Reach is seasonally variable, with levels near neutral (7.5 to 8.0 standard units [s.u.]) during the winter and increasing in the spring and summer (7.7 to 8.1 s.u.). Peak values (8.0 to 9.2 s.u.) are recorded during May and September (Raymond 2010). Longitudinally, pH ranges from 7.3 to 9.2 s.u. in this reach, with the lowest values recorded downstream from J.C. Boyle Reservoir and the highest values in Copco and Iron Gate reservoirs (Raymond 2008, 2009, 2010). In 2009, springtime pH levels at J.C. Boyle Reservoir were typically 8.5 s.u., decreasing during the summer and fall to 7.6 to 7.9 s.u. (Watercourse Engineering, Inc. 2011). As part of monthly sampling events between March and November from 2000 to 2005 and June through November 2007, pH was measured at Klamath River sites along within Iron Gate and Copco reservoirs (Figure C-38; PacifiCorp 2008c). Summer pH values tend to fluctuate the most at all sites with the largest variations and highest pH values occurring primarily between September and November upstream of Copco No. 1 Reservoir and between August and October downstream of Copco No. 1 Reservoir (PacifiCorp 2008c).

Depth profiles of pH in the Copco and Iron Gate reservoirs show pH generally decreasing with depth in both reservoirs (Table C-5; PacifiCorp 2008c). At depths less than 8 meters (the epilimnion), pH is greater than 8.5 s.u. in approximately 30 percent of samples (144 of 494) collected in Copco Reservoir and 20 percent of samples (25 of 485[4]) collected in Iron Gate Reservoir, while pH less than 7.0 s.u. occurred in less than 2 percent (6 of 494) of the samples. At depth greater than 20 meters (the hypolimnion), pH is greater than 8.5 s.u. in less than 1 percent of samples (1 of 391 in Copco No. 1 and 0 of 613 in Iron Gate) collected in both reservoirs, but approximately 17 percent of samples (68 of 391) collected in Copco Reservoir and 22 percent of samples (135 of 613) collected in Iron Gate Reservoir record pH values less than 7.0 s.u. (PacifiCorp 2008c). The distribution of pH values in both reservoirs is attributed to a phytoplankton response to nutrient inputs from upstream sources with photosynthesis in the upper reservoir waters (the epilimnion) resulting in higher pH levels (PacifiCorp 2008c; Raymond 2008). Subsequent analysis of pH data from 2005 to 2010 concurs with previous results that high pH near the surface during summer stratification is likely due to higher phytoplankton biomass and productivity from buoyant cyanobacteria [blue-green algae] near reservoir surfaces (Asarian and Kann 2011). Monthly water quality grab samples collected in Copco and Iron Gate reservoirs at three depths (near the surface [epilimnion], thermocline [metalimnion], and near the bottom [hypolimnion]) show similar pH trends in both reservoirs between 2011 and 2015 with pH usually between approximately 7.0 and 9.0 s.u., but ranging from 5.3 to 9.9 s.u. in Copco No. 1 Reservoir and 5.4 to 10.0 s.u. in Iron Gate Reservoir during 2013 (Watercourse Engineering, Inc. 2012, 2013, 2014, 2015, 2016).

Table C-5. Frequency of pH values in the Klamath River above or below threshold values from 2000 to 2007.

Location	Summary of pH values				
	N	N > 8.5	% > 8.5	N < 7.0	% < 7.0
Klamath River above Shovel Creek	72	7	9.7	1	1.4
Copco Reservoir	1202	148	12.3	84	7.0
Copco Reservoir < 8 m	494	144	29.1	6	1.2
Copco Reservoir > 18 m	391	1	0.3	68	17.4
Iron Gate Reservoir	1470	116	7.9	189	41.9
Iron Gate Reservoir < 8 m	485	25	19.6	8	1.6
Iron Gate Reservoir > 20 m	613	0	0.0	135	22.0
Below Iron Gate Dam	71	3	4.2	4	5.6
Klamath River at I-5	30	7	23.3	2	6.7
Klamath River near Shasta River	7	6	85.7	0	0.0

Source: PacifiCorp 2008c.

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[4] In PacifiCorp (2008c) Table 5.2-11, reproduced here as Table C-5, the number of samples with pH greater than 8.5 is listed as 25 of 485 total samples, but the percent of samples with pH greater than 8.5 is listed as 19.6 percent. The inconsistency between the number of samples with pH greater than 8.5 and the percent of samples with pH greater than 8.5 cannot be resolved with the available information in PacifiCorp (2008), so both are presented here for completeness and transparency
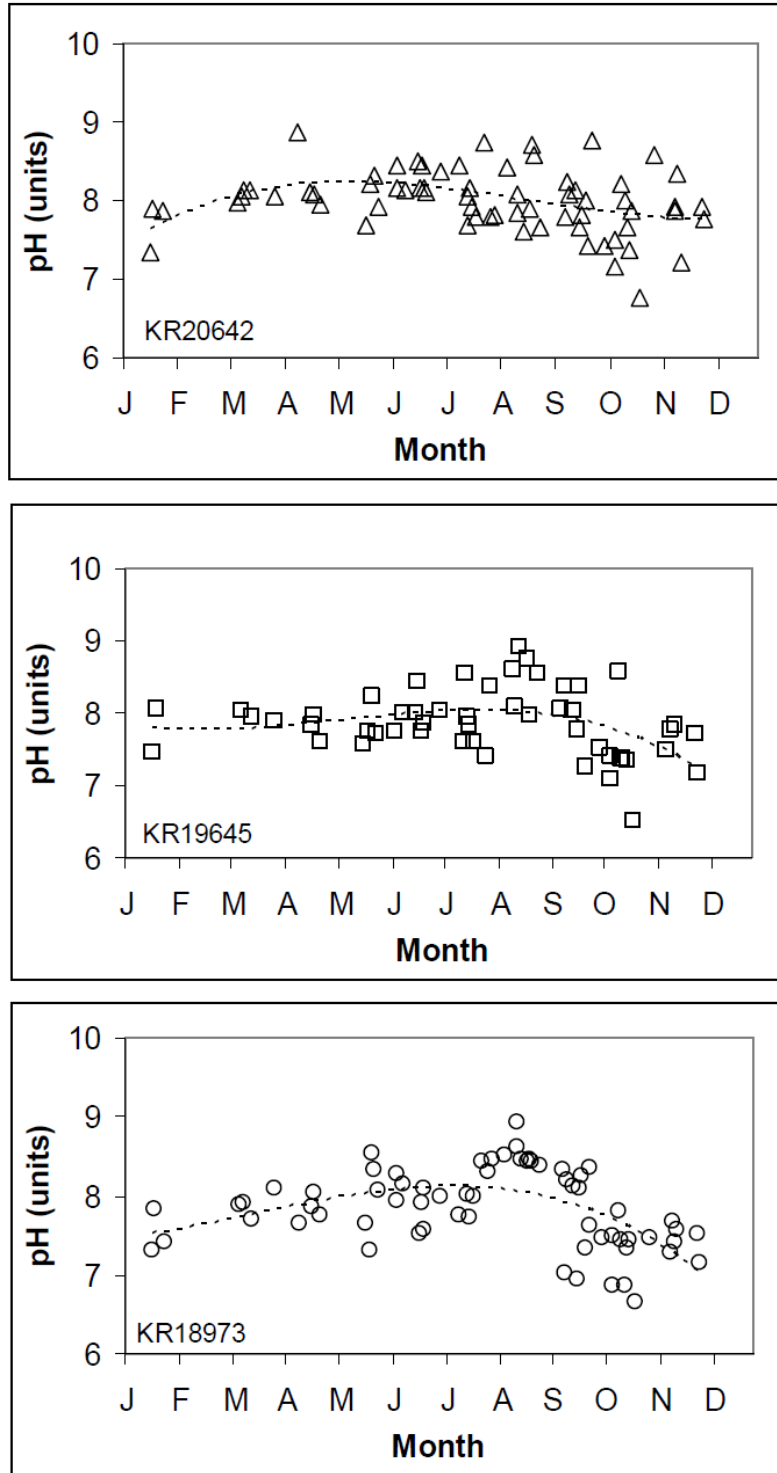
Figure C-38. Seasonal variations in pH values measured in the Klamath River upstream of Copco Reservoir near Shovel Creek (KR20642), downstream of Copco No. 2 powerhouse (KR19645), and downstream of Iron Gate Dam (KR18973). Source: PacifiCorp 2008c.

C.5.2   Mid- and Lower Klamath Basin

C.5.2.1   Iron Gate Dam to Salmon River

Downstream of Iron Gate Dam (RM 193.1) to Salmon River (RM 66.3) pH varies both longitudinally and temporally with the highest seasonal values generally occurring during late-summer and early fall months (August to September) depending on the location in the Klamath River (Figure C-39; Asarian and Kann 2013). The Basin Plan pH maximum of 8.5 s.u. is regularly exceeded in the Klamath River downstream from Iron Gate Dam (FISHPRO 2000; Karuk Tribe of California 2002, 2003, 2007, 2009, 2010a, 2010b, 2011, 2012, 2013; YTEP 2004, 2005; FERC 2007; USFWS 2008; North Coast Regional Board 2010, 2011; Watercourse Engineering, Inc. 2011a, 2011b, 2012, 2013, 2014, 2015, 2016; Asarian and Kann 2013). During April through October 2000 to 2009, incidences of pH *below* the minimum Basin Plan limit of 7.0 s.u. were also observed immediately downstream from Iron Gate Dam (RM 193.1), but pH below 7.0 s.u. was not recorded after 2009 (Karuk Tribe of California 2002, 2003, 2009, 2010a, 2010b, 2011, 2012, 2013; PacifiCorp 2004b; Watercourse Engineering, Inc. 2011a, 2011b, 2012, 2013, 2014, 2015, 2016). The Klamath River is a weakly buffered system (i.e., has typically low alkalinity below 100 mg/L as calcium carbonate [$CaCO_3$]), so it is susceptible to photosynthesis-driven daily swings in pH. Observed exceedances of pH water quality objectives usually occur during later afternoon or early evening, following the period of maximum photosynthesis (North Coast Regional Board 2010, Asarian and Kann 2013). A synthesis of pH measurements from 2001 to 2011 shows the daily range (i.e., daily maximum minus daily minimum) of pH generally peaks between late July and early September corresponding to daily cycles of photosynthesis and respiration (Figure C-38; Asarian and Kann 2013).
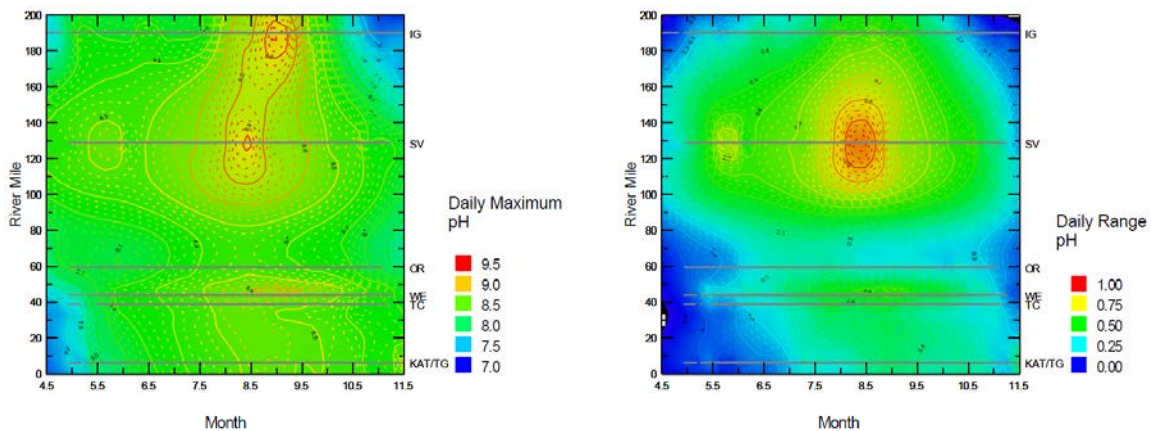
Figure C-39. Distribution of Isopleths Showing the 2001-2011 Average Daily Maximum pH and Daily Range of pH at Klamath River Sites Downstream of Iron Gate Dam from Around Mid-April (4.5) to Mid-November (11.5). Grey Lines Represent the Times the Dissolved Oxygen Concentration Was Measured in the Klamath River Downstream of Iron Gate Dam (IG at RM 193.1), at Seiad Valley (SV at RM 132.7), at Orleans (OR at RM 58.9), at Weitchpec Upstream of the Trinity River (WE at RM 43.6), Upstream of Tully Creek (TC at RM 40.1), and at Turwar (KAT/TG at RM 5.6). Source: Asarian and Kann 2013.

The most extreme pH exceedances typically occur from Iron Gate Dam (RM 193.1) to approximately Seiad Valley (RM 132.7) with pH values generally increasing from Iron

Gate Dam to the Klamath River upstream of Shasta River (RM 179.5) then decreasing with distance downstream (Figure C-40; FERC 2007; Karuk Tribe of California 2007, 2009, 2010a, 2010b, 2011, 2012, 2013; Asarian and Kann 2013). During May to October 2005, the greatest number of pH exceedances in this reach occur just upstream of the mainstem confluence with the Shasta River (RM 179.5) (Figure C-41). In 2007, daily maximum pH values downstream from Iron Gate Dam (RM 193.1) were 8.2 to 9.6 s.u. with the highest documented pH occurring in September (Figure C-42); near Seiad Valley (RM 132.7), maximum pH values were slightly lower, at 8.1 to 9.4 s.u. with the highest documented pH occurring in mid-August (Figure C-43; Karuk Tribe of California 2007). A synthesis of all the data from June through October 2001 to 2011 shows these trends are consistent over time with pH greater than 8.5 s.u. in 23 percent of measurements downstream of Iron Gate Dam, 35 percent of measurements upstream from the Shasta River, and 26 percent of measurements at Seiad Valley. Measurements of pH greater than 8.5 s.u. are most frequent in August and September for most mainstem locations, but they are most frequent in July and August in the Klamath River upstream of Shasta River. Measurements of pH greater than 9.0 s.u. are most frequent at Iron Gate Dam (9 percent for September), upstream from Shasta River (8 percent for August), and Seiad Valley (6 percent for August), but measurements of pH greater than 9.0 s.u. are rare (less than 0.1 percent) at mainstem locations downstream of Seiad Valley. High daily maximum pH values at Iron Gate Dam are correlated with high chlorophyll-*a* concentrations from upstream algal blooms rather than large 24-hour cycles (Asarian and Kann 2013).

Trends in pH from 2011 to 2015 are generally similar to previous findings with pH initially increasing with distance downstream of Iron Gate Dam before then decreasing with distance downstream to Salmon River, but there is substantial variability in the pH both seasonally and during individual days (Watercourse Engineering, Inc. 2012, 2013, 2014, 2015, 2016). Measurements of pH in 2015 demonstrate typical patterns with the seasonal range in pH being greater immediately downstream of Iron Gate Dam, but the daily range in pH being greater at Seiad Valley (Figure C-44; Watercourse Engineering, Inc. 2016). Downstream of Iron Gate Dam, pH ranges from approximately 7.5 to 9.5 s.u., while pH ranges from approximately 7.5 to 9.0 s.u. at Seiad Valley.
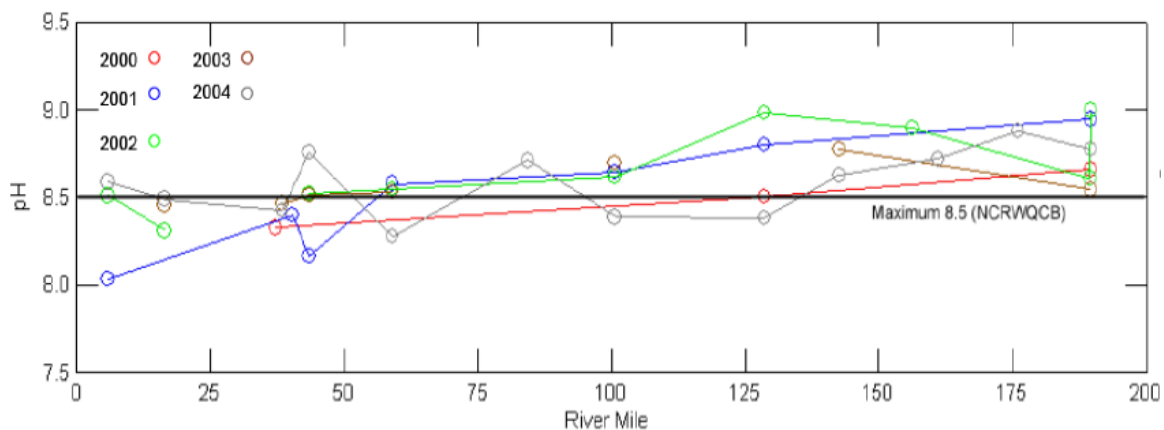
Figure C-40. Average August Daily Maximum pH Values for Locations along the Mainstem Klamath River Downstream from Iron Gate Dam for the Years 2000 to 2004 using Data Collected by USFWS, USGS, and the Karuk Tribe of California and Yurok Tribe. Source: Kier Associates 2006 as cited in FERC 2007.
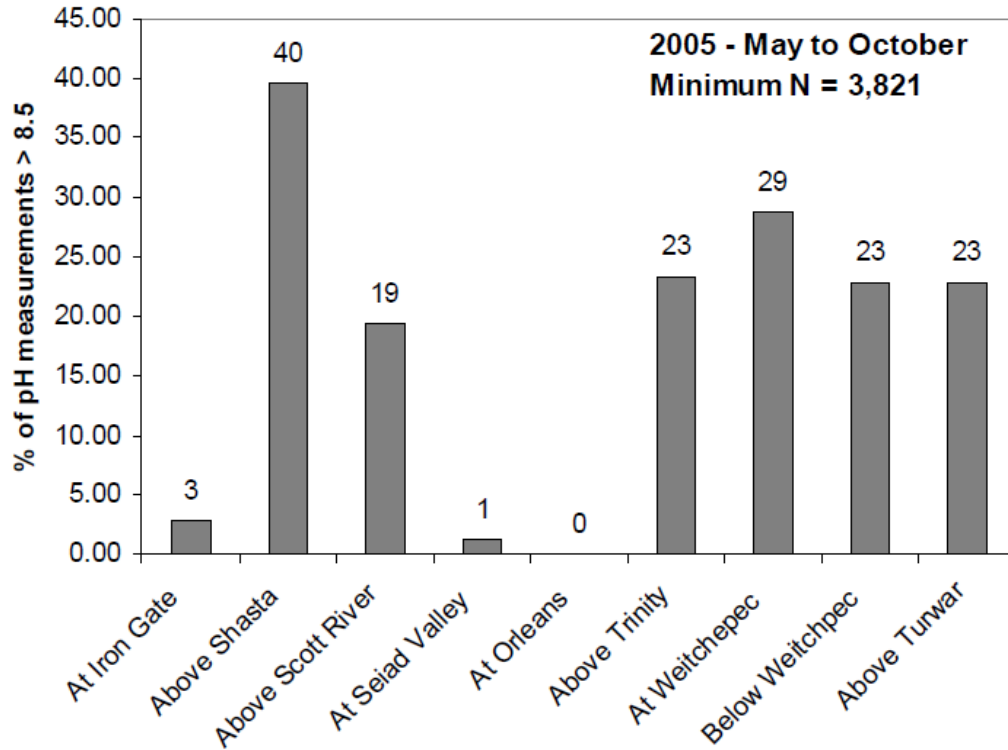
Figure C-41. Percent of pH Measurements in the Lower Klamath River Exceeding the Basin Plan Water Quality Objective of 8.5 s.u. during 2005. Source: North Coast Regional Board 2010.
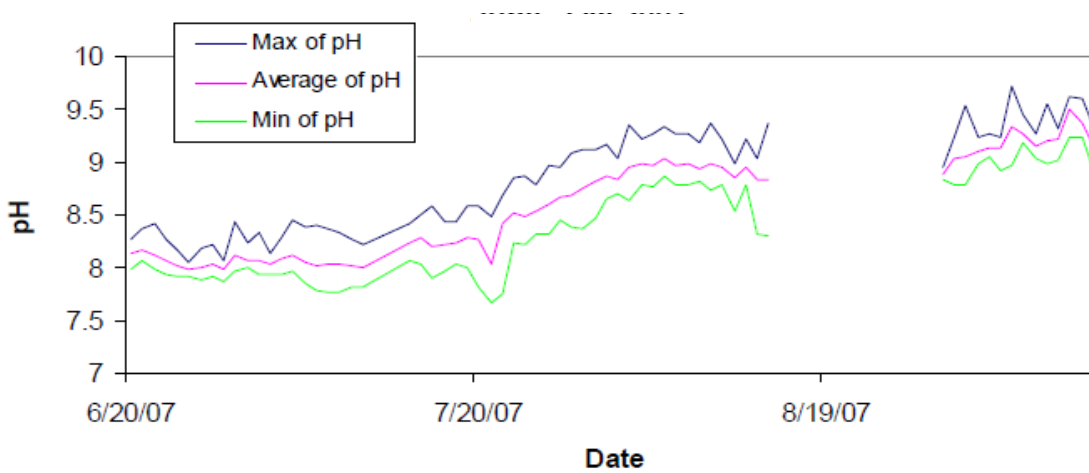
Figure C-42. Daily Maximum, Mean, and Minimum pH in the Klamath River Downstream from Iron Gate Dam (RM 193.1) from June to October 2007. Source: Karuk Tribe of California 2007.
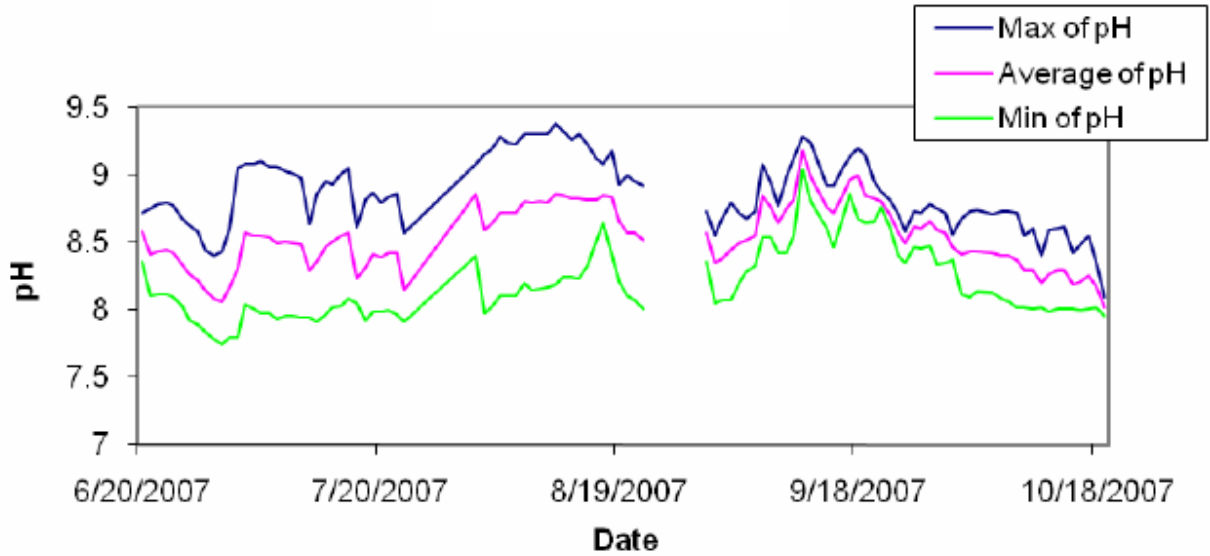
Figure C-43. Daily Maximum, Mean, and Minimum pH in the Klamath River near Seiad (≈RM 132.7) from June to October 2007. Source: Karuk Tribe of California 2007.
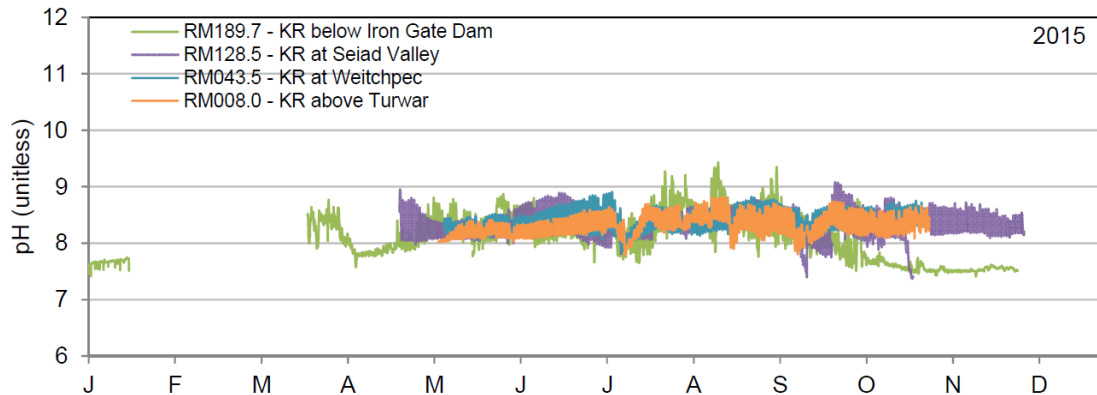
Figure C-44. Continuous pH Data from 2015 in the Klamath River Downstream of Iron Gate Dam (RM 193.1), at of Seiad Valley (RM 132.7), at Weitchpec (RM 43.6), and Upstream of Turwar (RM 5.6). Source: Watercourse Engineering, Inc. 2016.

C.5.2.2   Salmon River to Estuary

The Basin Plan pH maximum of 8.5 s.u. is also regularly exceeded in the lower Klamath River between the Salmon River (RM 66.3) and Turwar Creek (RM 5.6) during summer months with most of the exceedances occurring downstream of the Trinity River (Figures C-39 and C-41; FISHPRO 2000; Karuk Tribe of California 2002, 2003, 2009, 2010a, 2010b, 2011, 2012, 2013; YTEP 2004, 2005; USFWS 2008; North Coast Regional Board 2010, 2011; Sinnott 2010a, 2011a, 2012a; Watercourse Engineering, Inc. 2011a, 2011b, 2012, 2013, 2014, 2015, 2016; Asarian and Kann 2013 Hanington 2013; Hanington and Ellien 2013; Hanington and Cooper Carouseli 2014). Water quality monitoring by the Karuk Tribe includes pH data from Orleans (RM 58.9), which is just downstream from the mainstem confluence with the Salmon River (see also Section

C.5.2.1). Daily maximum pH values at Orleans were 7.9 to 8.9 s.u. from June through October 2007, with the highest pH occurring in mid-September (Figure C-45; Karuk Tribe of California 2007). Analysis of data from 2001 to 2011 by Asarian and Kann (2013) shows similar trends, yet pH at Orleans most frequently exceeds 8.5 s.u. during August (19 percent) rather than September (12 percent). More contemporary data from 2012 and 2013 further indicate that the maximum pH and frequency of exceeding 8.5 s.u. varies between August and September, depending on the year, with pH peaking in August in 2012 and peaking in September in 2013 (Karuk Tribe of California 2012, 2013).

In the mainstem river between the confluence with the Trinity River and the Klamath River Estuary, annual water pH monitoring has been conducted by the YTEP since 2002 (YTEP 2004, 2005; Sinnott 2010a, 2011a, 2012a; Watercourse Engineering, Inc. 2011a, 2011b, 2012, 2013, 2014, 2015, 2016; Asarian and Kann 2013; Hanington 2013; Hanington and Ellien 2013; Hanington and Cooper Carouseli 2014). The pH trends downstream of Weitchpec (RM 43.6) to Turwar (RM 5.6) vary seasonally with a general decreasing trend during the earlier part of the year then during September through October the pH decreases from the Klamath River at Weitchpec (RM 43.6) to the Klamath River upstream of Tully Creek (RM 40.1) then increases from upstream of Tully Creek downstream to Turwar (Figure C-39; Asarian and Kann 2013). The timing of peak pH and highest frequency of exceeding 8.5 s.u. varies between July and September depending on the year. Between 2001 and 2011, the pH most frequently exceeds 8.5 s.u. in September at Weitchpec (36 percent) and at Turwar (24 percent), while it most frequently exceeds 8.5 s.u. in the Klamath River upstream of Tully Creek (RM 40.1) in August (23 percent) (Asarian and Kann 2013). However, during 2009 monitoring, peak pH values were documented from July through September with the highest daily maximums recorded in early July. The highest pH values were documented at the most upstream location (i.e., just over 9.0 s.u. at Klamath River at Weitchpec [RM 43.6]), while both sample locations farther downstream were approximately 8.8 s.u. (Klamath River upstream of Tully Creek [RM 40.1] and upstream of Turwar Boat Ramp [RM 6]) (Figure C-46; Sinnott 2010a). More recent pH monitoring from 2012 and 2013 also show the variability in timing of pH peaks with pH at Weitchpec (RM 43.6) peaking at 8.7 s.u. in October in 2012 and 8.8 s.u. in July in 2013. The pH recorded upstream of the Turwar Boat Ramp (RM 6) is more consistent with pH peaking at approximately 8.7 s.u. during August in both 2012 and 2013 (Hanington 2013; Hanington and Ellien 2013).
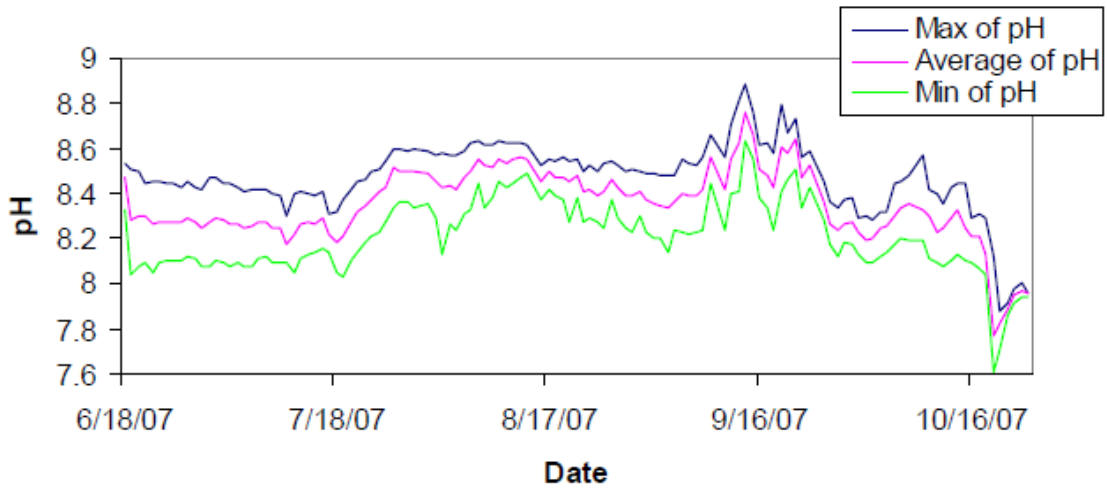
Figure C-45. Daily Maximum, Mean, and Minimum pH on the Klamath River near Orleans (RM 58.9) from June to October 2007. Source: Karuk Tribe of California 2007.
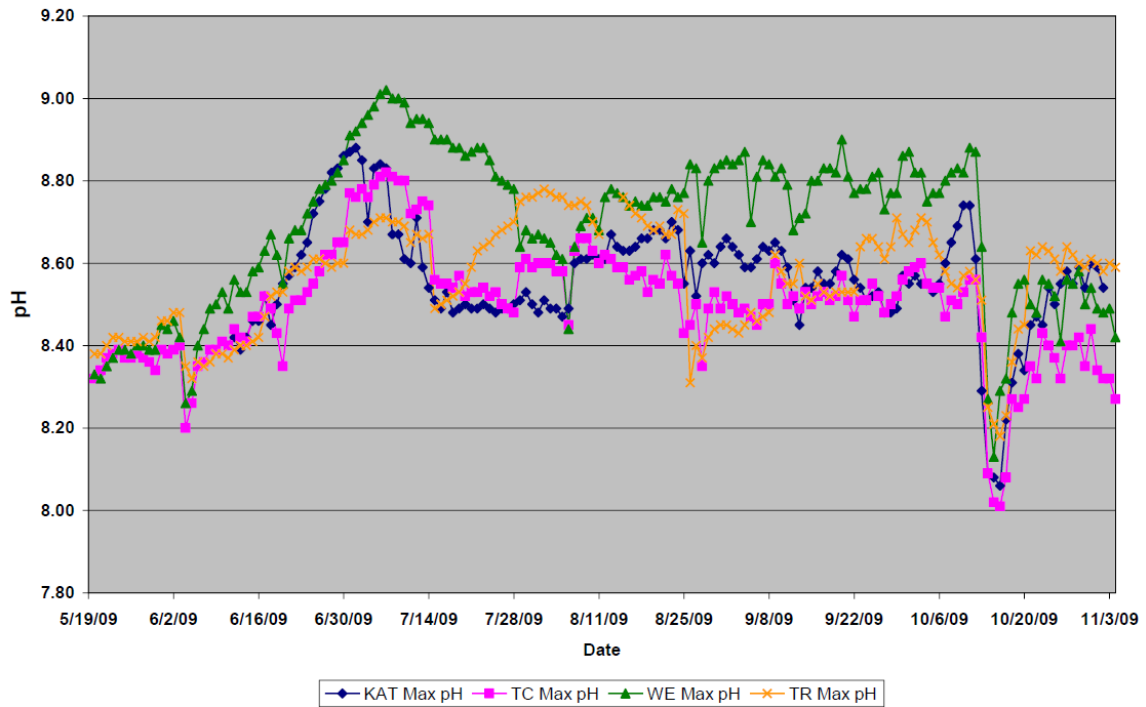
Figure C-46. Daily Maximum pH in the Klamath River at Weitchpec (RM 43.6 [WE]), Upstream of Tully Creek (RM 40.1 [TC]), and Upstream of Turwar Boat Ramp (RM 6 [KAT]), as well as in the Trinity River (RM 43.3 [TR]) near the Confluence with the Klamath River (RM 0.5 [TR]). Source: Sinnott 2010a.

C.5.2.3   Klamath River Estuary

pH within the Klamath River Estuary is variable spatially and temporally and is influenced by season, river flow, vertical stratification (thermal and/or salinity), and location of the estuary mouth, the latter changing due to periodic sand bar movement. The Basin Plan pH maximum of 8.5 s.u. is regularly exceeded in the Klamath River Estuary though pH measured during monthly grab samples near the surface of the lower estuary from 2009 to 2015 only show pH exceeding 8.5 s.u. in February 2011 and August 2013 (YTEP 2005; Sinnott 2010a, 2011a, 2012a; Watercourse Engineering, Inc. 2011a, 2011b, 2012, 2013, 2014, 2015, 2016; Hanington and Torso 2013; Hanington and Stawasz 2014; Hanington and Cooper Carouseli 2014). Based on Yurok Tribe water quality data, pH in the Klamath Estuary ranges between approximately 6.9 to 9.0 s.u. though values below 7.0 s.u. are occasionally measured, with peak values generally occurring during the summer months (YTEP 2005; Sinnott 2010a, 2011a, 2012a; Watercourse Engineering, Inc. 2011a, 2011b, 2012, 2013, 2014, 2015, 2016; Hanington and Torso 2013; Hanington and Stawasz 2014; Hanington and Cooper Carouseli 2014). Daily variations in pH are typically on the order of 0.5 s.u., and fluctuations tend to be somewhat larger in the late summer and early fall. The Environmental Fluid Dynamics Code estuary model component used in the California Klamath River TMDL development as well as observed data show very low phytoplankton and chlorophyll-*a* concentrations in the Klamath River Estuary (YTEP 2005), suggesting that local photosynthesis and biological respiration are not significant enough to cause large daily fluctuations of pH, as seen in upstream reaches. When large daily fluctuations are observed, they are likely caused by an upstream daily signal that is subsequently transported into the Klamath River Estuary.

C.6   Algal Toxins and Chlorophyll-*a*

C.6.1   Upper Klamath Basin

C.6.1.1   Hydroelectric Reach

Seasonal phytoplankton blooms (also called algal blooms) in the Hydroelectric Reach have been recorded historically, with chlorophyll-*a* concentrations in Iron Gate Reservoir ranging from 0.3 micrograms per liter (ug/L) to 21.6 ug/L during March, July, and October 1975 (USEPA 1978). Over the past decade, algal toxins and chlorophyll-*a* have become routinely monitored water quality parameters in the Hydroelectric Reach. PacifiCorp's chlorophyll-*a* monitoring data for the river upstream of J.C. Boyle Reservoir to immediately downstream from Iron Gate Dam from 2002 through 2009 (May to October) indicates that annual mean values above 10.0 ug/L are typical of the dataset and there is generally greater apparent variability upstream of J.C. Boyle Reservoir as compared with just downstream from Iron Gate Reservoir (PacifiCorp 2004a; Raymond 2008, 2009, 2010). Chlorophyll-*a* concentrations have a wider range with higher peak values in reservoirs compared to the free-flowing portion of the Klamath River (Raymond 2008; Asarian and Kann 2011).

A broader longitudinal analysis of measured chlorophyll-*a* concentrations was conducted using monitoring data compiled during 2005 to 2007 (May to September) from the Yurok Tribe, Karuk Tribe of California, North Coast Regional Board, and PacifiCorp (North Coast Regional Board 2010). Results at numerous locations from the lower Klamath River Estuary (RM 0 to 3.9) to J.C. Boyle Dam (RM 229.8) demonstrate that median chlorophyll-*a* concentrations within Copco No. 1 and Iron Gate reservoirs are 2 to 10

times greater (note the logarithmic scale in Figure C-47) than those documented in free-flowing locations in the mainstem river, with median concentrations greater than 10.0 ug/L exhibited in the reservoirs and median concentrations less than 10.0 ug/L exhibited at river locations (North Coast Regional Board 2010). Analysis of chlorophyll-*a* measurements from June to October 2005 to 2010 show similar trends with chlorophyll-*a* higher in Copco No. 1 and Iron Gate reservoirs than in river locations upstream, between, and downstream of the reservoirs (Figure C-48; Asarian and Kann 2011). Chlorophyll-*a* concentrations upstream of Copco No. 1 Reservoir ("Abv Shovel" location in Figure C-47; "KRACorig" location in Figure C-48) are lower than those measured downstream within the reservoir suggesting chlorophyll-*a* concentrations in the reservoirs are due to algal blooms originating in the reservoirs. Chlorophyll-*a* concentrations are generally higher in Copco No. 1 Reservoir than Iron Gate Reservoir though annual variability does result in Iron Gate Reservoir having higher median chlorophyll-*a* concentrations during some years (Asarian and Kann 2011). In the Copco No. 1 and Iron Gate reservoirs, median chlorophyll-*a* concentrations are highest near the surface than decrease with depth (Figure C-48). Chlorophyll-*a* data indicates concentrations at a 5 m depth are elevated compared to inflow values from upstream with chlorophyll-*a* in Copco No. 1 (and Iron Gate to a lesser extent) exceeding 10.0 ug/L for much of August to September in 2005 to 2007 (Asarian and Kann 2011). Peak chlorophyll-*a* concentrations indicating periods of algal blooms are generally larger in Copco No. 1 than Iron Gate with some exceptions (Asarian and Kann 2011).

Seasonal chlorophyll-*a* patterns in the Hydroelectric Reach indicate that relatively high concentrations can occur during spring diatom blooms (e.g., approximately 30.0 to 40.0 ug/L for Copco No. 1 and Iron Gate reservoirs in March 2000 to 2003), followed by a period of relatively low concentrations after the blooms die (e.g., less than 10 mg/L for Copco No. 1 and Iron Gate reservoirs in April to July 2000 to 2003). In some years (e.g., 2009 and 2010), the intense spring blooms have included the blue-green algae *Anabaena* spp. with sufficient density to require health advisory posting of the reservoirs. A second increase occurs during August and September when dense blooms dominated by both *Aphanizomenon flos-aquae* and *Microcystis aeruginosa* are typical (e.g., approximately 30.0 to 58.0 ug/L for Copco No. 1 and Iron Gate reservoirs 2000 to 2003) (FERC 2007). Asarian and Kann (2011) found similar seasonal chlorophyll-*a* trends in 2005 to 2010 data with peak values occurring from March to April and August to September. Chlorophyll-*a* concentrations from November to April were lower downstream of Iron Gate Dam than upstream of Copco due to settling of diatoms from upstream (Asarian and Kann 2011).
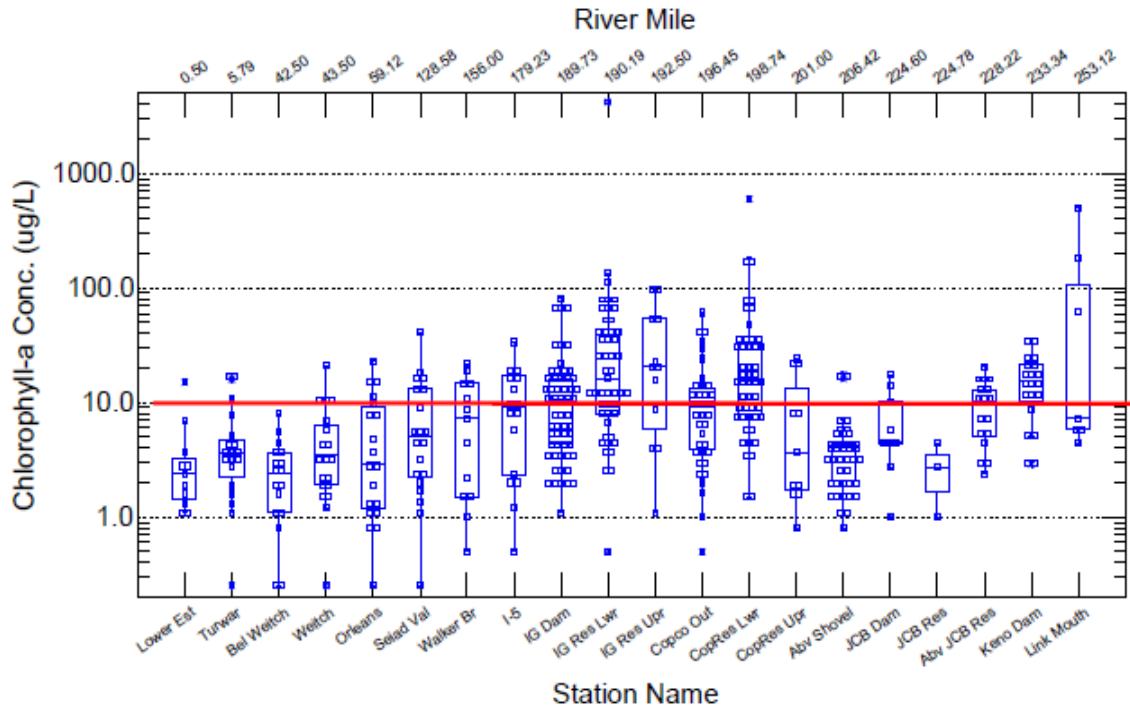
Figure C-47. Longitudinal Analysis of Summer (May through September) Chlorophyll-*a* Concentrations from 2005 to 2007 Along the Klamath River. Note the Logarithmic Scale. River miles specified for Klamath River features are based on those accurate at the time of the report and differ slightly from 2018 river mile designations (Table 3.2-1). Data from the Yurok Tribe, Karuk Tribe of California, North Coast Regional Water Quality Control Board, and PacifiCorp. Source: North Coast Regional Board 2010.
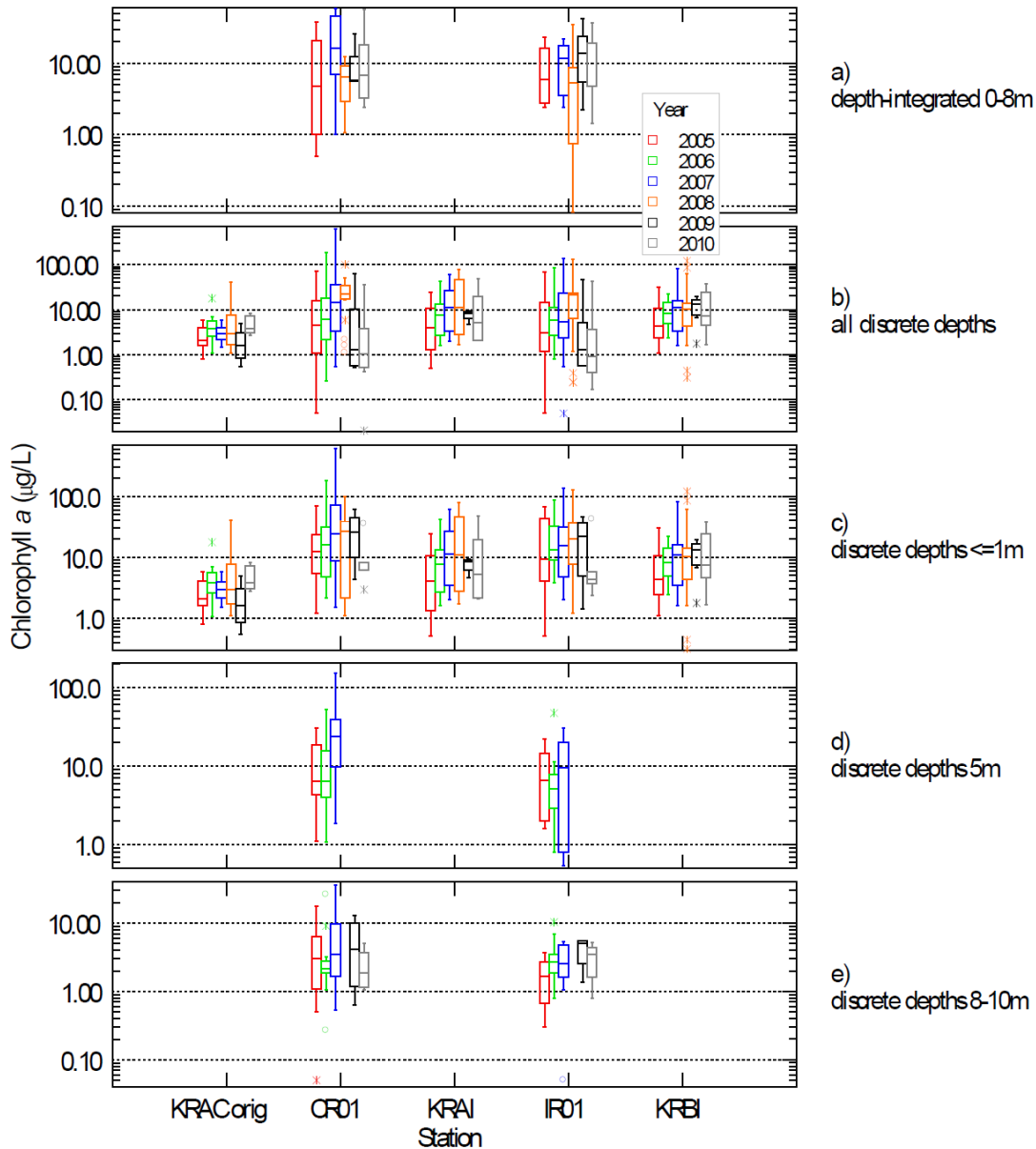
Figure C-48. Longitudinal and Vertical Chlorophyll-*a* Concentrations June to October 2005 to 2010 at the Klamath River Upstream of Copco (KRACorig), Copco Reservoir Before the Dam (CR01), Klamath River Upstream of Iron Gate (KRAI), Iron Gate Reservoir Before the Dam (IR01), and Klamath River Downstream of Iron Gate Dam (KRBI). Source: Asarian and Kann 2011.

Algal toxins, also referred to as cyanotoxins, can be produced by some species of blue-green algae, which are also called cyanobacteria, especially during large seasonal blooms in reservoir or lake environments. Cyanotoxins (e.g., cyclic peptide toxins such as microcystin that act on the liver, alkaloid toxins such as anatoxin-a and saxitoxin that act on the nervous system) can cause irritation, sickness, or in extreme cases, death to exposed organisms, including humans (WHO 1999). Species capable of producing microcystin include *Microcystis aeruginosa*, while species in the genus *Anabaena* can produce anatoxin-a and saxitoxin. More complete listings of specific toxins produced by

genera of blue-green algae worldwide are provided in Lopez et al. (2008) and ODEQ (2011). The California Cyanobacteria and Harmful Algal Bloom (CCHAB) Network, a multi-agency workgroup formerly called the Statewide Blue-Green Algae Working Group, has developed guidance for responding to harmful algal blooms (HABs), cyanotoxin [algal toxin] threshold levels for protection of human health, and posting and de-posting cyanotoxin [algal toxin] triggers in recreational waters (State Water Board et al. 2010, updated 2016). The State Water Resources Control Board (State Water Board), the California Department of Public Health (CDPH), and the California Environmental Protection Agency (CalEPA) Office of Environmental Health and Hazard Assessment (OEHHA) Caution Action trigger threshold for the protection of human health in recreational waters was previously 8 ug/L of microcystin, but it is 0.8 ug/L of microcystin in the 2016 update (State Water Board et al. 2010, updated 2016).

Microcystin concentrations are generally low from J.C. Boyle to Copco No. 1 reservoirs, higher between Copco No. 1 and Iron Gate reservoirs, then decrease with distance downstream from Iron Gate Dam (Kann 2006; Jacoby and Kann 2007; Kann 2007a, b, c, d; Kann and Corum 2007, 2009; CH2M Hill 2008; Kann et al. 2010a; Watercourse Engineering, Inc. 2011a, 2011b, 2012, 2013, 2014, 2015, 2016; Otten et al. 2015). High chlorophyll-*a* concentrations have been shown to correlate with the toxigenic blue-green algae blooms dominated by *Anabaena* spp. and *Microcystis aeruginosa* and sharp increases in microcystin levels above WHO numeric targets (Kann and Corum 2009) and the State Water Board, the CDPH, and CalEPA OEHHA public health thresholds (State Water Board et al. 2010, updated 2016). Data collected from 2005 through 2009 indicate high levels of microcystin in Copco No. 1 and Iron Gate reservoirs, with measured concentrations exceeding the current State Water Board, CDPH, and CalEPA OEHHA public health threshold of 0.8 ug/L for microcystin by over 10,000 times in Copco No. 1 Reservoir in 2006, 2007, 2008, and 2009 and by over 1,000 times in Iron Gate Reservoir in 2006 and 2009 (Figure C-49; Jacoby and Kann 2007; Kann 2007a, b, c, d; Kann and Corum 2007, 2009; Kann et al. 2010a).

Microcystin measured during May to December 2009 exhibited extremely high concentrations (1,000 to 73,000 ug/L) during algal blooms occurring in July, August, and September in Copco No. 1 Reservoir in Mallard Cove and Copco Cove, and in Iron Gate Reservoir at Jay Williams. The highest microcystin concentration (73,000 ug/L) was measured on September 28, 2009 at Mallard Cove in Copco No. 1 Reservoir. Microcystin concentrations in Iron Gate Reservoir at Jay Williams peaked in October at 3,200 ug/L (Watercourse Engineering, Inc. 2011a). More contemporary measurements of microcystin in the Copco No. 1 and Iron Gate reservoirs during May to December 2015 exhibit very high concentrations consistent with previous findings (Watercourse Engineering, Inc. 2016). In Copco No. 1 Reservoir at Mallard Cove and Copco Cove, microcystin increases in June to July 2015 until peaking at 12,000 to 16,000 ug/L during algal blooms while microcystin concentrations also increase to between 200 to 370 ug/L in August, September, and October. In Iron Gate Reservoir at Jay Williams, microcystin concentrations in 2015 peaked at 770 ug/L in September, while microcystin in Iron Gate Reservoir at Camp Creek reached a maximum of 64 ug/L (Watercourse Engineering, Inc. 2016).
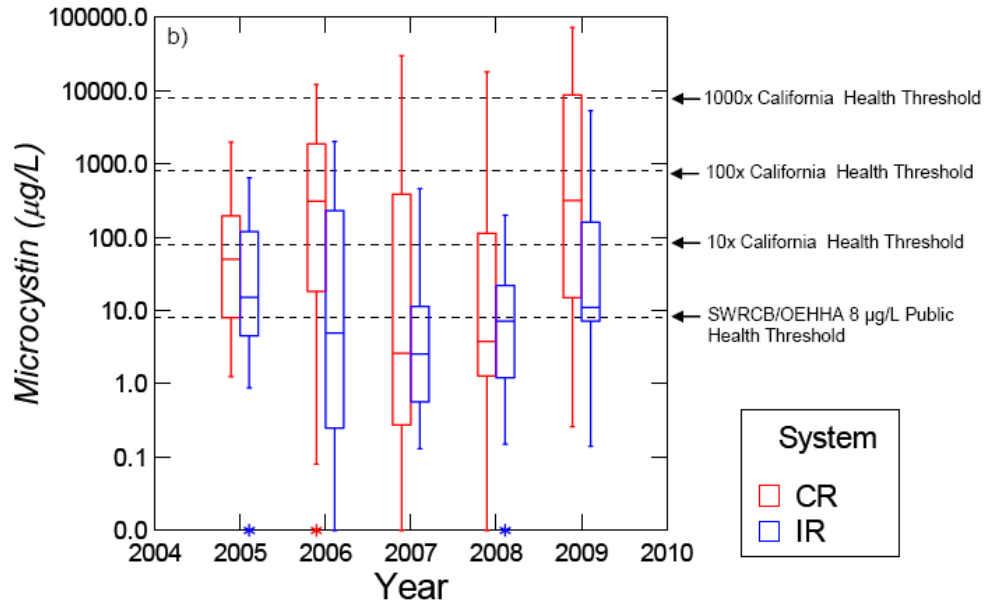
Figure C-49. Inter-annual Comparison of Microcystin Concentration for Copco No. 1 Reservoir (Red Square) and Iron Gate Reservoir (Blue Square) during July through October 2005 to 2009. Note: the SWRCB [State Water Board]/[CalEPA] OEHHA Public Health Threshold for microcystin was 8 ug/L in 2010, but it was revised to 0.8 ug/L in 2016. Source: Kann et al. 2010a.

Otten et al. (2015) report the longitudinal and temporal variations in microcystin levels from upstream of Copco No. 1 to Turwar indicate Iron Gate Reservoir is the principal source of *Microcystis* cells contributing to microcystin concentrations downstream of Iron Gate Dam. The consistently low microcystin levels above Copco No. 1 along with phycological (conditions associated with algae), genetic, and toxin analyses that identify different population dynamics in Copco No. 1 and Iron Gate reservoirs indicate microcystin in the Copco No. 1 and Iron Gate reservoirs is due to local *Microcystis aeruginosa* populations (Otten et al. 2015). In 2012, Copco No. 1 microcystin concentrations peak in August then decline while Iron Gate Reservoir microcystin concentrations peak in September with the temporal differences in peak microcystin occurring because depth differences between the reservoirs cause thermal stratification in Iron Gate approximately two weeks later than Copco No. 1. The timing of peak microcystin levels in Iron Gate corresponds to the highest microcystin concentrations in the Klamath River downstream of Iron Gate Dam consistent with Iron Gate Reservoir being the source of downstream *Microcystis aeruginosa* cells. Genetic analysis of the *Microcystis aeruginosa* populations showed Copco No. 1 Reservoir populations were dominated by one genetic type the entire year, but the populations in Iron Gate Reservoir and immediately downstream of Iron Gate Dam had a simultaneous switch in the dominant genetic type in late August that was also detected in the further downstream populations. The simultaneous timing of the genetic change in Iron Gate Reservoir and downstream *Microcystis aeruginosa* populations, but no corresponding genetic change in Copco No. 1 Reservoir provides direct evidence that downstream populations are originating in Iron Gate Reservoir rather than Copco No. 1 (Otten et al. 2015).

In 2007, a *Microcystis aeruginosa* bloom prompted a Yurok Tribe health advisory along multiple affected reaches in the Klamath River (Kann 2007a, b, c, d); data from July through September 2007 also indicate microcystin bioaccumulation in fish and mussel tissue samples collected in the Klamath River and Iron Gate and Copco No. 1 reservoirs (Kann 2008a; see Section 3.3.3.3 for more information on algal toxins in fish and mussel tissue). Additional public health advisories were issued in 2009 and 2010 in Copco No. 1 and Iron Gate reservoirs, as well as downstream locations in the Klamath River (including locations on the Yurok Reservation), for microcystin levels in ambient and/or freshwater mussel tissue (Fetcho 2010, Kann et al. 2010a, Kann et al. 2010b). Data from 2008 and 2009 did not show microcystin bioaccumulation in the tissue and liver samples from fish collected from Copco No. 1 and Iron Gate reservoirs (CH2M Hill 2009, PacifiCorp 2010a). In 2010, the Lower Klamath Project reservoirs were posted to protect public health due to elevated blue-green algae cell counts and algal toxin (i.e., microcystin) concentrations. In 2013, 2014, and 2016, public health advisories were posted for both Copco No. 1 and Iron Gate reservoirs when algal toxin levels or blue-green algae cell concentrations exceeded one or more public health advisory threshold (North Coast Regional Board 2013, 2014, 2016). Copco No. 1 Reservoir also had a public health advisory posted in July 2017 when microcystin concentrations exceeded both the "caution action trigger" threshold (0.8 ug/L) and the "warning, Tier 1" threshold (6 ug/L) (North Coast Regional Board 2017) with microcystin concentrations reaching 380,000 ug/L in Copco No. 1 Reservoir in August 2017 (E&S Environmental Chemistry, Inc. 2017).

As part of an evaluation of the relationship between *Microcystis aeruginosa* cell density and microcystin concentration, Kann et al. (2010a) compared the 2009 measured values to the 2010 WHO guidelines for a low probability of adverse health effect (20,000 cells/mL *Microcystis aeruginosa*, or 4 ug/L microcystin) and the 2010 State Water Board/ [CalEPA] OEHHA guidelines for protection against a moderate probability of adverse effects (40,000 cells/mL *Microcystis aeruginosa*, or 8 ug/L microcystin). These results showed that the more conservative guideline of 20,000 cells/mL *Microcystis aeruginosa* decreases the frequency of exceeding the 8 ug/L 2010 State Water Board/ [CalEPA] OEHHA guideline value for microcystin, and it is more protective of public health (Kann and Corum 2009, Kann 2014). Overall, the 2005 to 2012 results clearly illustrate that the majority of exceedances to all guidelines and thresholds occurred in the reservoirs in the Hydroelectric Reach (as compared with downstream riverine sites), with the highest overall levels measured in Copco No. 1 Reservoir (Figure C-50; Kann and Corum 2009, Kann and Bowman 2012, Kann 2014, Watercourse Engineering, Inc. 2016). Concentrations of microcystin toxin in Iron Gate and Copco No. 1 reservoirs are typically 1 to 3 orders of magnitude greater relative to the Klamath River downstream of Iron Gate Dam (Raymond 2008, Kann et al. 2010a, Kann and Bowman 2012, Kann 2014, Watercourse Engineering, Inc. 2016). Overall, the available data indicate that while river exceedances to the 2010 microcystin water quality guidelines do occur, they are far less in number than exceedances in Copco No. 1 and Iron Gate reservoirs (Figure C-50; see also Raymond 2008, Kann et al. 2010a, Kann and Bowman 2012, Kann 2014, Watercourse Engineering, Inc. 2016).
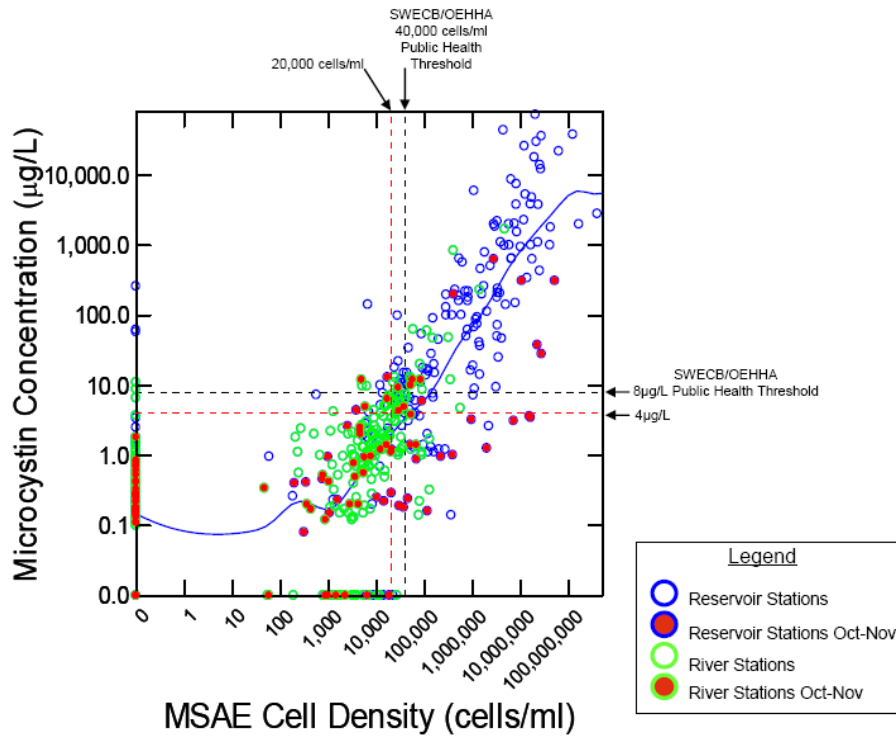
Figure C-50. Relationship between Microcystis aeruginosa Cell Density and Microcystin Toxin Concentration for Copco No. 1 and Iron Gate Reservoirs and Klamath River Stations 2005 to 2009. Note: the SWRCB [State Water Board]/[CalEPA] OEHHA Public Health Threshold for microcystin was 8 ug/L in 2010, but it was revised to 0.8 ug/L in 2016. Similarly, the SWRCB [State Water Board]/[CalEPA] OEHHA Public Health Threshold for Microcystis aeruginosa Cell Density was 40,000 cells/mL in 2010, but it was revised to 4,000 cells/mL of any toxin producing blue-green algae in 2016. Source: Kann et al. 2010a.

Anatoxin-a produced by blue-green algae species in the genus *Anabaena* was detected in Iron Gate Reservoir on September 3, 2005 in testing by the California Department of Health Services (Kann 2007b; Kann 2008b). In additional monitoring conducted for the Karuk Tribe, no anatoxin-a was detected during 2005, 2006, 2007, 2008 in Copco No. 1 or Iron Gate reservoirs (Kann and Corum 2006, 2007, 2009; Kann 2007d). While concentrations of *Anabaena flos-aquae* cells have continued to be monitored, anatoxin-a concentrations are not available for Lower Klamath Project reservoir sites in recent years.

C.6.2   Mid- and Lower Klamath Basin

C.6.2.1   Iron Gate Dam to Salmon River

As noted above (Section C.6.1.1), 2005 to 2007 data indicate that during May through September median chlorophyll-*a* concentrations decrease longitudinally with distance downstream from Iron Gate Dam (Figure C-47) to the Klamath River confluence with the Salmon River (RM 66.3), but concentrations remain greater than those measured just upstream of Copco No. 1 Reservoir. This suggests that algal blooms occurring in the reservoirs were being transported to the downstream river reaches. Ward and Armstrong (2010) report the 2001 to 2005 mean chlorophyll-*a* concentrations ranged

from approximately 0.5 to 3.5 ug/L, with concentrations generally decreasing with distance downstream from Iron Gate Dam though variability is observed (Figure C-41). During 2001 to 2005, the highest annual mean value (approximately 5.0 ug/L) occurs in 2005 at the confluence with the Shasta River (RM 179.5). In 2009, the Karuk Tribe collected chlorophyll-*a* and pheophytin-*a* (an additional photosynthetic pigment) data from the Klamath River downstream from Iron Gate Dam; chlorophyll-*a* values were approximately 1.0 to 35.0 ug/L and were variable depending on location. Generally speaking, relatively greater values were observed at upstream locations near Iron Gate Dam (RM 193.1) and Walker Bridge sites, but the peak value was observed farther downstream of the confluence with the Salmon River (RM 66.3) at Orleans (RM 58.9) (Karuk Tribe of California 2010a). Analysis of the chlorophyll-*a* data from 2001 to 2011 by Asarian and Kann (2013) summarizes the June to October annual variability and range of chlorophyll-*a* concentrations downstream of Iron Gate Dam (Figure C-52). The highest values occur in 2007 and 2008 with chlorophyll-*a* concentrations greater than 100 ug/L downstream of Iron Gate Dam, yet most measurements were less than 50 ug/L. Chlorophyll-*a* concentrations have a seasonal variation with concentrations in Seiad Valley (RM 132.7) decreasing in May/June, then increasing from July until peaking in late August/early September, and finally declining from September to October (Asarian and Kann 2013). Monthly/bi-weekly monitoring data from 2012 to 2015 show similar trends with chlorophyll-*a* concentrations generally decreasing from downstream of Iron Gate Dam (RM 193.1) to the Klamath River downstream of Seiad Valley (RM 132.7), but chlorophyll-*a* concentrations occasionally remained approximately the same or increased between Iron Gate Dam and Seiad Valley (Watercourse Engineering, Inc. 2013, 2014, 2015, 2016). In June to October 2015, chlorophyll-*a* concentrations range from approximately 1.3 to 9.5 ug/L downstream of Iron Gate Dam (RM 193.1), while chlorophyll-*a* concentrations range from approximately 1.3 to 9.6 ug/L downstream of Seiad Valley (RM 132.7) (Watercourse Engineering, Inc. 2016).
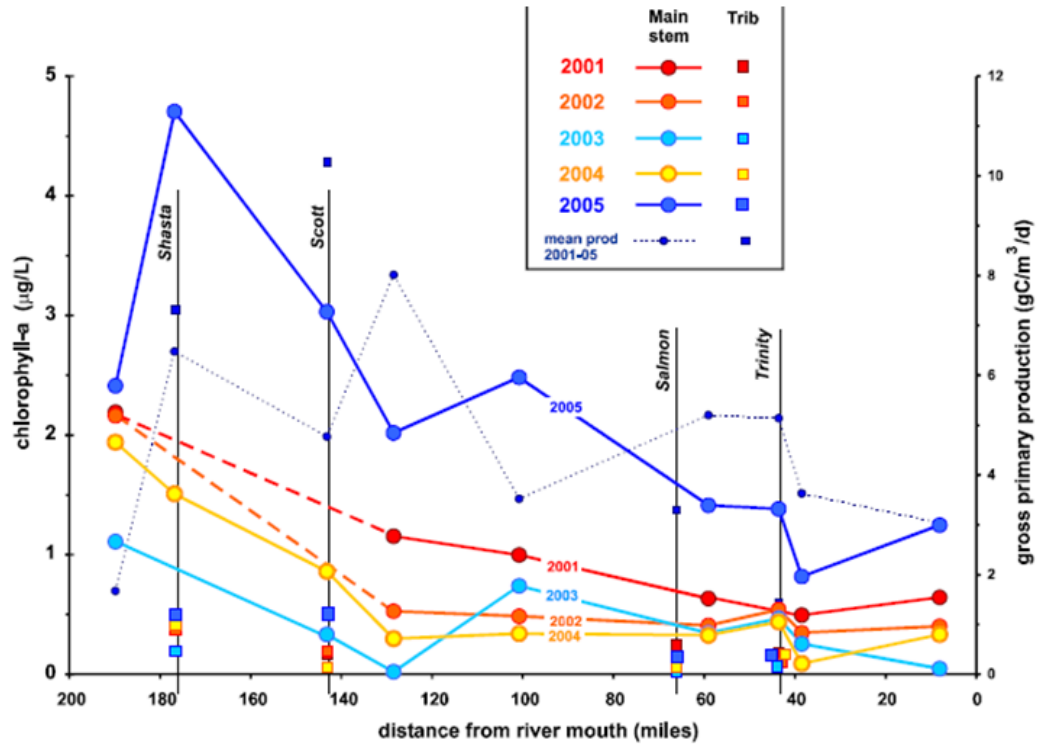
Figure C-51. Annual mean values of chlorophyll-*a* in the Klamath River downstream from Iron Gate Dam during June to September 2001 to 2005. River miles specified for Klamath River features are based on those accurate in 2010 and differ slightly from 2018 river mile designations (Table 3.2-1). Source: Ward and Armstrong 2010.
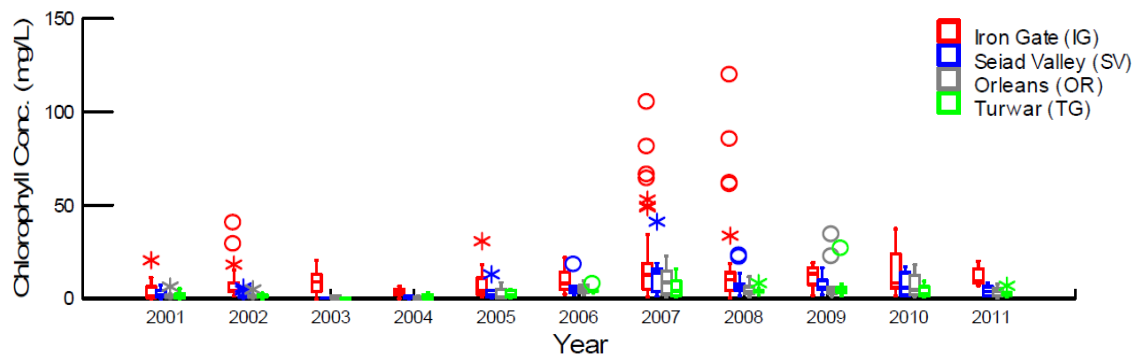
Figure C-52. Chlorophyll-*a* in the Klamath River downstream from Iron Gate Dam during June to October 2001 to 2011. Source: Asarian and Kann 2013.

In 2008, the Karuk Tribe collected blue-green algae concentrations (cells/mL) using optical phycocyanin probes to allow more timely assessment of public health threats from toxigenic blue-green algae species. Data from downstream from Iron Gate Dam collected during June to October indicated peak values (greater than 25,000 cells/mL) in July and early-to-mid September (Karuk Tribe of California 2009). Blue-green algae

concentrations (cells/mL) were also collected by the Karuk Tribe in 2016 at sites downstream of Iron Gate Dam to the confluence with the Salmon River with peak values (greater than 30,000 cells/mL) occurring in July (Watercourse Engineering, Inc. 2016).

Although concentrations of both *Microcystis aeruginosa* and microcystin in the Klamath River downstream from the Hydroelectric Reach are lower relative to the reservoirs (Figures C-53 and C-54), WHO guidelines for exposure to microcystin (i.e., less than 4.0 ug/L) have been exceeded downstream from Iron Gate Dam on numerous occasions (Kann 2004; Kann and Corum 2009; Kann et al. 2010a; Fetcho 2010; Watercourse Engineering, Inc. 2011a, 2011b, 2012, 2013, 2014, 2015, 2016; KTWQC 2016), including late-summer/early-fall *Microcystis aeruginosa* blooms in September 2007, 2009, 2010, 2011, 2012, 2013, and 2016 from Iron Gate Dam (RM 193.1) to the mouth of the Klamath River (RM 0.0). Health Advisories were posted along the Iron Gate Dam to Shasta River reach of the Klamath River in 2009 and 2010, due to elevated *Microcystis aeruginosa* cell counts and/or microcystin concentrations in river water. During 2009, mean microcystin concentrations immediately downstream from Iron Gate Dam (RM 193.1) were approximately 2.0 ug/L, with mean values decreasing to less than 1.0 ug/L at sites further downstream to approximately Orleans (RM 58.9) (Watercourse Engineering, Inc. 2011a). However, two measurements exceeded 7.0 ug/L at stations located at RM 156 and 128.5 (Watercourse Engineering, Inc. 2011a). Baseline grab sample water quality monitoring between 2012 and 2015 demonstrate the range of annual variability in longitudinal microcystin concentration trends (Figure C-54). Additional public health sampling, separate from the baseline monitoring, in 2015 show microcystin concentrations peaking in late July to early August between 2.9 and 5.6 ug/L (Watercourse Engineering, Inc. 2016). Microcystin concentrations collected in 2012 and 2013 document how microcystin concentrations vary over a 24-hour period and the potential influence of grab sampling time on microcystin measurements. On September 17 to 18, 2013, microcystin downstream of Iron Gate Dam ranges from approximately 8.0 to 16.0 ug/L with minimum values occurring around noon and the peak occurring at midnight (Kann 2014).

Available data indicate that algal blooms in Iron Gate and Copco No. 1 reservoirs have been responsible for the public health exceedances in the lower river (Kann 2006, Kann and Corum 2009). The highest microcystin concentrations in the Klamath River downstream of Iron Gate Dam coincide with peak microcystin levels in Iron Gate Reservoir. This is consistent with the reservoir being the source of downstream *Microcystis* cells causing elevated microcystin concentrations. A simultaneous genetic change in Iron Gate Reservoir and downstream *Microcystis aeruginosa* populations, but no corresponding genetic change in Copco No. 1 Reservoir provides further evidence that downstream populations are originating in Iron Gate Reservoir rather than Copco No. 1 (Otten et al. 2015).

Additionally, data from 2007 indicate microcystin bioaccumulation in juvenile salmonids reared in Iron Gate hatchery (Kann 2008a) and, in 2010, algal toxins were found in salmonid tissues collected near Happy Camp (Kann et al. 2011, Kann et al. 2013) (see also Section 3.3.3.3).
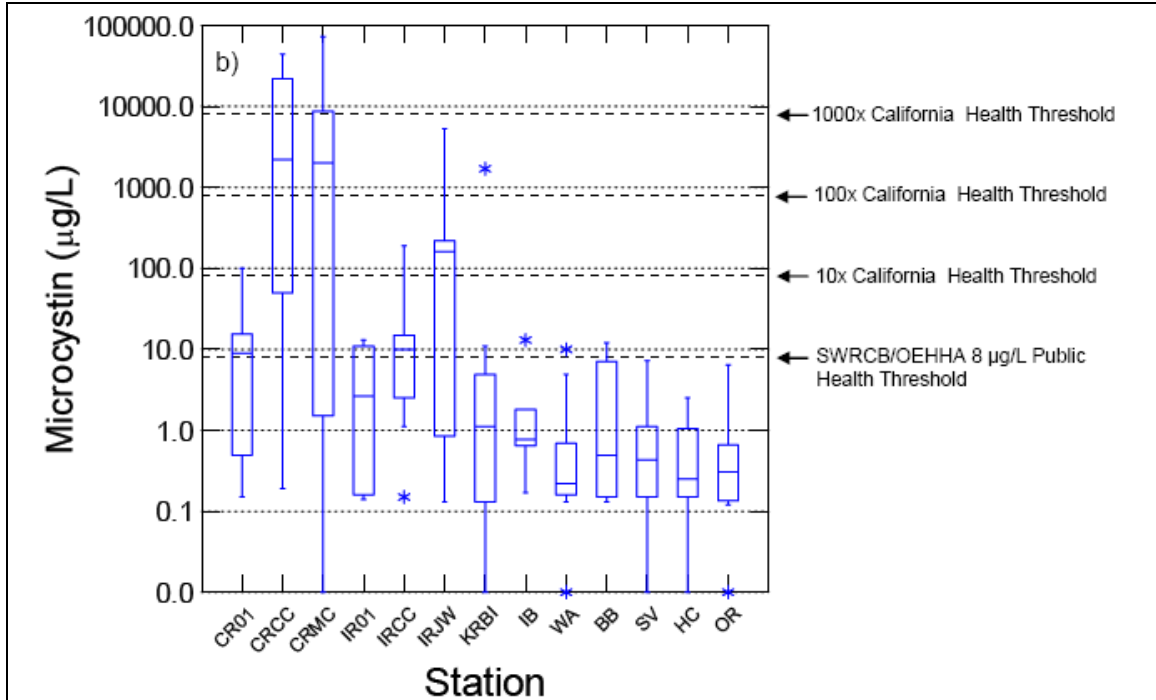
Figure C-53. Microcystin Concentration in Klamath River from Copco No. 1 (CR01) to Orleans (RM 58.9) during June to November 2009. WA=Walker Bridge, SV=Seiad Valley (RM 132.7), OR=Orleans (RM 58.9). Note: the SWRCB [State Water Board]/[CalEPA] OEHHA Public Health Threshold for microcystin was 8 ug/L in 2010, but it was revised to 0.8 ug/L in 2016. Source: Kann et al. 2010a.
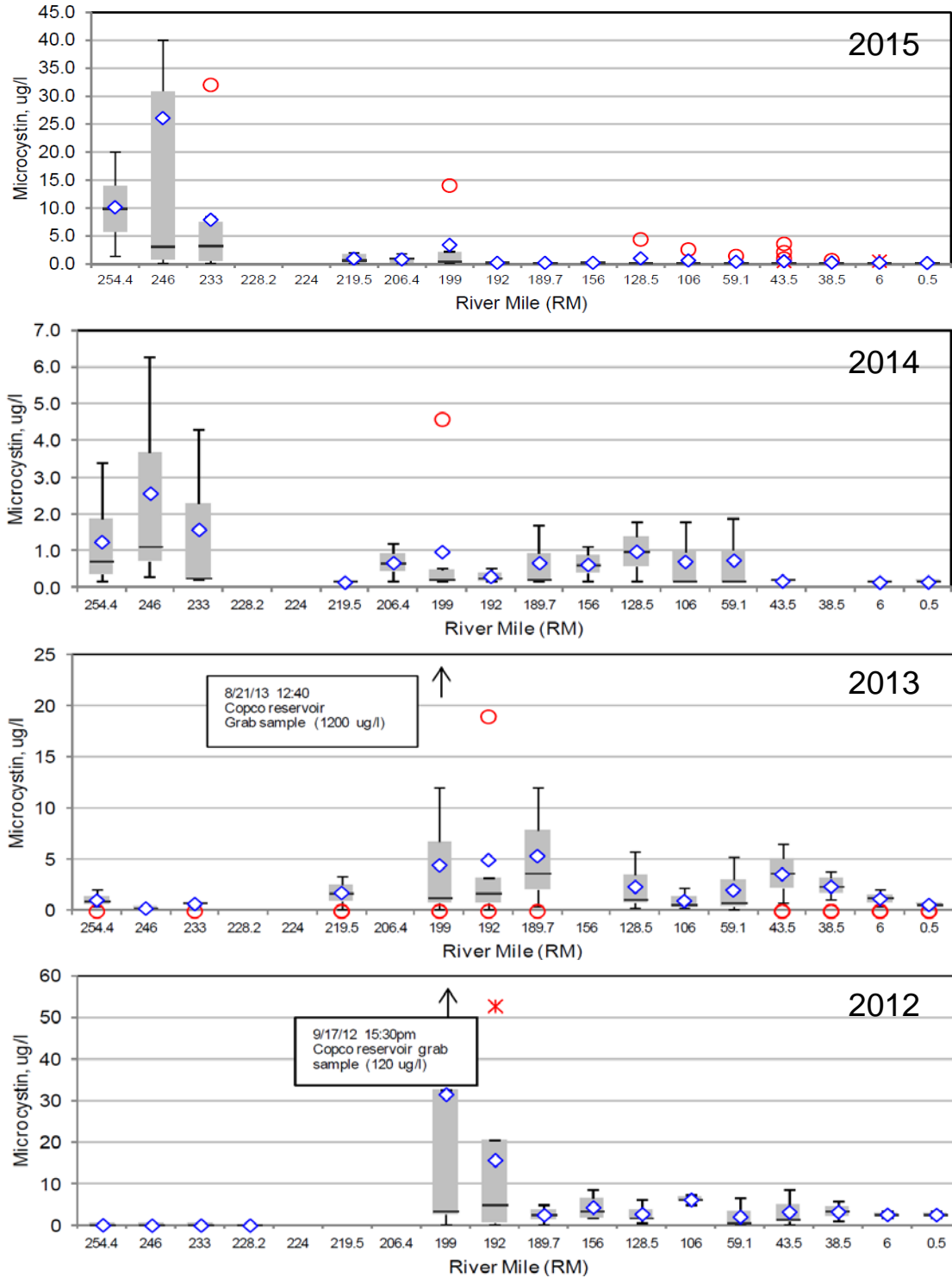
Figure C-54. Klamath River Microcystin Concentration trends from measurements made between February and December with median (-), mean (◊), outlier (*), and extreme outliers (○) identified. River miles specified are based on those accurate at the time of the reports and differ slightly from 2018 river mile designations (Table 3.2-1). Source: Watercourse Engineering, Inc. 2013, 2014, 2015, 2016.

C.6.2.2   Salmon River to Estuary

Downstream from the confluence with the Salmon River (RM 66.3), chlorophyll-*a* and algal toxin concentrations exhibit variability and are generally lower than those measured farther upstream. Water quality monitoring from 2008 to 2012 at Saints Rest Bar (RM 44.9) in the Klamath River shows chlorophyll-*a* usually ranging from approximately 1.0 to 15.0 ug/L, but occasional higher peaks with one peak reaching 44.9 ug/L in August 2011 (HVTEPA 2013). The Yurok Tribe monitors chlorophyll-*a*, pheophytin-*a*, and blue-green algae concentrations (cells/mL) annually, and use optical phycocyanin probes to allow more timely assessment of public health threats from toxigenic blue-green algae species on the Yurok Reservation (YTEP 2005; Fetcho 2006, 2007, 2008, 2009, 2011; Sinnott 2008, 2009a, 2009b, 2010b, 2011b, 2012b; Watercourse Engineering, Inc. 2011a, 2011b, 2012, 2013, 2014, 2015, 2016; Hanington and Torso 2013; Hanington and Stawasz 2014; Hanington and Cooper Carouseli 2014; Gibson 2016). Chlorophyll-*a* 2001 to 2005 data from Ward and Armstrong (2010) show small relative increases in chlorophyll-*a* with distance downstream, from near the Trinity River confluence (RM 43.3) to Turwar (RM 5.6), (Figure C-51), suggesting that phytoplankton productivity may increase slightly as water moves toward the Klamath River Estuary. Chlorophyll-*a* data measured from Weitchpec (RM 43.6) to the Klamath River Estuary during 2003 to 2004 did not show that trend though with concentrations, where detectable, generally decreasing in the downstream direction and below 5.0 ug/L (YTEP 2004, 2005). During 2006 to 2010 from May to October, chlorophyll-*a* concentrations frequently show an increase in chlorophyll-*a* with distance downstream from the confluence with the Trinity River (RM 43.3) to the Turwar Boat Ramp (RM 6) consistent with the findings of Ward and Armstrong (2010) (Sinnott 2008, 2009a, 2009b, 2010b, 2011b). Between May and July chlorophyll-*a* is typically less than 5.0 ug/L, but it increases during August to October with peak concentrations of 24.0 to 27.0 ug/L in 2009 (Sinnott 2008, 2009a, 2009b, 2010b, 2011b). Longitudinal chlorophyll-*a* concentrations from 2011 to 2014 are variable from the confluence with the Trinity River (RM 43.3) to the Turwar Boat Ramp (RM 6) with concentrations seasonally increasing or decreasing with distance downstream (Sinnott 2012b; Hanington and Torso 2013; Hanington and Stawasz 2014; Hanington and Cooper Carouseli 2014). In 2011 to 2014, chlorophyll-*a* is less than 6.0 ug/L from May to July then increases to approximately 10.0 to 15.0 ug/L during August to October before decreasing to below 3.0 ug/L in December (Sinnott 2012b; Hanington and Torso 2013; Hanington and Stawasz 2014; Hanington and Cooper Carouseli 2014). The peak values between 2006 and 2014 usually occur at Weitchpec (RM 43.6) from late August to mid-October and varying by year (6.7 ug/L in 2008 and 26.0 ug/L in 2009), but there is one exception in November 2013 when chlorophyll-*a* at Weitchpec reaches 27.0 ug/L.

During 2009, mean microcystin concentrations from Orleans (RM 58.9) to Klamath River at Klamath (RM 5.9) were less than 1.0 ug/L, well below the 2010 State Water Board/ [CalEPA] OEHHA guideline value for microcystin (8 ug/L), but greater than the 2016 State Water Board/ [CalEPA] OEHHA guideline for microcystin (0.8 ug/L) (Watercourse Engineering, Inc. 2011a). Individual microcystin measurements generally remained less than 1.0 ug/L as well, with the exception of a sample collected in late-September at Orleans (RM 58.9) for which the microcystin concentration was 6.4 ug/L (Watercourse Engineering, Inc. 2011a). Microcystin at Saints Rest Bar (RM 44.9) in 2010 to 2012 usually ranges from below 0.15 ug/L (the detection limit) to 5.6 ug/L, but microcystin in September 2012 reaches a peak of 19.0 ug/L (HVTEPA 2013). Microcystin concentrations from 2010 to 2015 range from less than 1.0 ug/L to approximately

12.0 ug/L with concentrations usually highest at Weitchpec (RM 43.6) then decreasing with distance downstream (Gibson 2016) though microcystin was occasionally higher at Orleans (RM 58.9) than Weitchpec (RM 43.6) in some years (Figure C-54). Microcystin concentrations peak between July and September around the same time the maximum concentration of *Microcystis aeruginosa* are measured, but Gibson (2016) notes microcystin is present at high levels in some samples even when *Microcystis aeruginosa* are not. While peak microcystin in the Klamath River at Weitchpec exceeds 8.0 ug/L in 2010, 2012, and 2013, microcystin concentrations downstream of Weitchpec, are less than 8.0 ug/L for all years between 2010 and 2015 (Gibson 2016). Microcystin exceeded 0.8 ug/L at Weitchpec at least once every year between 2010 to 2015, but microcystin exceeded 0.8 ug/L only once every year in 2010 through 2013 in the Klamath River at the Turwar Boat Ramp (RM 6). While microcystin levels in fish tissue samples are usually below detection level or trace (0.17 micrograms per gram [ug/g]), 0.54 ug/g microcystin was measured in a half pounder steelhead liver collected in the Klamath River at Weitchpec in 2005 (Kann 2006).

As described for the Klamath River from Iron Gate Dam to the Salmon River (Section C.6.2.1), there have been numerous exceedances of public health guidelines in the Klamath River from the Salmon River confluence to the Klamath River Estuary, particularly in 2010. Public health advisories were issued in 2009 and 2010 in this reach (including locations on the Yurok Reservation) for elevated microcystin levels in ambient and/or freshwater mussel tissue samples (Fetcho 2010; Kann et al. 2010a; Kann et al. 2010b). In addition, substantial bioaccumulation (exceeding public health guidelines) of microcystin in freshwater mussels has been shown in this reach (Kann 2008a, Kann et al. 2010b). In 2014 and 2015, public health advisories were posted when microcystin concentrations exceeded the 0.8 ug/L Recommended Threshold for Recreational Waters and blue-green algae cell concentrations were elevated (YTEP 2014, 2015).

Anatoxin-a was not detected above the reporting limit in water samples collected during 2008 and 2009 at Lower Klamath River monitoring sites (Fetcho 2009, 2011). In recent years, anatoxin-a has been measured in the Klamath River downstream of Iron Gate Reservoir on several occasions, typically in the lower reaches including at monitoring sites near Weitchpec and Orleans (Otten 2017). Concentrations of *Anabaena flos-aquae* cells have continued to be monitored in recent years, but anatoxin-a concentrations are not available for Klamath River monitoring sites.

C.6.2.3   Klamath River Estuary

Chlorophyll-*a* and algal toxin levels in the Klamath River Estuary are generally similar to those measured at stations just upstream. During 2006 to 2014, chlorophyll-*a* concentrations in the Klamath River Estuary were less than 5.0 ug/L from May to July (except for one measurement of 9.9 ug/L in July 2012), then ranged from 0.5 to 15.0 ug/L from August to October (Sinnott 2008, 2009a, 2009b, 2010b, 2011b, 2012b; Hanington and Torso 2013; Hanington and Stawasz 2014; Hanington and Cooper Carouseli 2014). Peak concentrations of chlorophyll-*a* during 2006 to 2014 occurred during late-July to mid-October and varied by year (2.4 ug/L in 2006, 15.0 ug/L in 2007).

Algal toxin concentrations in the Klamath River Estuary are generally below 4.0 ug/L, corresponding to relatively low concentrations of *Microcystis aeruginosa* with several exceptions (Fetcho 2006, 2007, 2008, 2009, 2011; Watercourse Engineering, Inc. 2011a, 2011b, 2012, 2013, 2014, 2015, 2016; Gibson 2016). In September 2007 and

2010 the Yurok Tribe issued advisories because *Microcystis aeruginosa* concentrations exceeded 40,000 cells/mL. In 2010 and 2011 measured microcystin concentrations exceeded 4.0 ug/L between September and October (Gibson 2016). Microcystin in the Klamath River Estuary (RM 0.5) exceeded 0.8 ug/L at least once every year in 2010 through 2012, but it did not exceed 0.8 ug/L in 2013 through 2015 (Gibson 2016). In one additional instance, in September 2005, concentrations exceeded the WHO guideline for low risk recreational use (20,000 cells/mL). These elevated levels of *Microcystis aeruginosa* corresponded with elevated levels measured farther upstream in Copco No. 1 and Iron Gate reservoirs (Kann and Corum 2006). Lastly, there is emerging evidence that algal toxins flushing from coastal rivers into Monterey Bay, California were responsible for numerous sea otter deaths in 2007 (Miller et al. 2010). While it is not known if conditions in Monterey Bay are similar to those in the Klamath River marine nearshore environment, there may be potential for microcystin to adversely impact marine organisms when large blooms are transported through the Klamath River Estuary and into the Pacific Ocean.

C.7   Inorganic and Organic Contaminants

C.7.1   Upper Klamath Basin

C.7.1.1   Hydroelectric Reach

Water Column Contaminants
Existing water quality data are available from the California Surface Water Ambient Monitoring Program (SWAMP), which collected water quality data, including inorganic and organic contaminant data, from 2001 through 2005 at eight monitoring sites from the Oregon-California state line (RM 214.1) to Klamath River at Klamath Glen (RM 5.9) (North Coast Regional Board 2008). Results from the state line site indicated that for the majority of inorganic constituents, excluding nutrients (i.e., arsenic, cadmium, chloride, chromium, copper, lead, mercury, nickel, selenium, silver, sulfate, and zinc), concentrations were in compliance with water quality objectives at the time of sampling. Nutrients are discussed above in Section C.3.1. Aluminum concentrations (50.7 to 99.2 ug/L) exceeded the USEPA continuous concentration for freshwater aquatic life protection (87 ug/L) on two of four site visits (50 percent exceedance rate), and exceeded the USEPA secondary Maximum Contaminant Level (MCL) for drinking water (50 ug/L) on all four site visits (100 percent exceedance rate) (North Coast Regional Board 2008). Grab samples were analyzed for 100 pesticides, pesticide constituents, isomers, or metabolites; 50 polychlorinated biphenyls (PCBs) congeners; and 6 phenolic compounds. Results indicated no PCBs detections, but one detection of dichlorodiphenyldichloroethylene (1,1-bis-(4-chlorophenyl)-2,2-dichloroethene or DDE) (25 percent of samples) and one detection of trans-nonachlor (25 percent of samples) were found (North Coast Regional Board 2008).

Sediment Contaminants
To investigate the potential for toxicity of the sediments trapped behind the Lower Klamath Project reservoirs, Shannon & Wilson, Inc. (2006) collected 25 cores from J.C. Boyle, Copco No. 1, and Iron Gate reservoirs during 2006 and analyzed them for contaminants including acid volatile sulfides, metals, pesticides, chlorinated acid herbicides, PCBs, volatile organic compounds (VOCs), semi-volatile organic compounds (SVOCs), cyanide, and dioxins. The locations of the sediment cores were distributed throughout each reservoir, including locations on the historical Klamath River channel

(on-thalweg) and the surrounding submerged terraces or near tributary mouths (off-thalweg) along the edge of the historical Klamath River. Four locations on-thalweg were sampled in J.C. Boyle Reservoir, with maximum core depths ranging from 0.3 feet at the upstream end of the reservoir to 13.2 feet near the dam. Twelve locations (7 on-thalweg, 5 off-thalweg) were sampled in Copco No. 1 Reservoir, with maximum core depths ranging from 1.5 feet at the upstream end of the reservoir to 12.1 feet near the middle of the reservoir. Nine locations (5 on-thalweg, 4 off-thalweg) were sampled in Iron Gate Reservoir, with maximum core depths ranging from 0.7 feet at the upstream end of the reservoir to 7.8 feet within the Slide Creek/Camp Creek arm of the reservoir. During sediment core drilling, the sediments were evaluated to distinguish recent reservoir-deposited sediment from pre-reservoir sediment with drilling logs noting the depth of different sediment horizons. Interval composite/depth interval sediment samples were generated from the sediment cores, including both the reservoir-deposited and pre-reservoir sediments, with the number of interval samples depending on the total depth of the sediment core. No herbicides or PCBs were found above U.S. Army Corps of Engineers (USACE) Puget Sound Dredged Disposal Analysis Program (PSDDA) screening levels and only one sample exceeded applicable PSDDA screening levels for VOCs ethyl benzenes and total xylenes (Shannon & Wilson, Inc. 2006). While cyanide was detected in two of three sediment cores, it was not found in toxic free cyanide form (HCN or CN$^-$), and it is not likely to be bioavailable or result in adverse effects on fish and other aquatic biota.

Dioxin, a known carcinogen, was measured in three samples from J.C. Boyle, Copco No. 1, and Iron Gate reservoirs. Long-term exposure to dioxin in humans is linked to impairment of the immune system, the developing nervous system, the endocrine system and reproductive functions. In the 2006 reservoir samples, measured levels were 2.48 to 4.83 pg/g (picograms per gram or parts per trillion [ppt] expressed as Toxic Equivalent Concentrations)) and did not exceed USACE (1,000 pg/g), proposed freshwater sediment Apparent Effects Threshold for benthic fauna (8.8 pg/g), International Joint Commission for Great Lakes Science Advisory Board (10 pg/g), PSDDA bioaccumulation (15 pg/g), or U.S. Environmental Protection Agency fish and wildlife guidelines (2.5 to 210 pg/g) screening levels (Shannon & Wilson, Inc. 2006). More comprehensive reviews of dioxin guidelines and sediment studies from watersheds outside of the Klamath basin were conducted by Dillon (2008) and USEPA (2010), the latter presenting an estimate of background dioxin concentrations (2 to 5 ppt) for non-source-impacted sediments throughout the U.S. and specifically in the western U.S. (USEPA 2010). Based on the information presented in (USEPA 2010), in addition to being within the range of natural background, Klamath dioxin sediment levels reported by Shannon & Wilson, Inc. (2006) are one to three orders of magnitude below risk-based USEPA (1,000 pg/g dry weight [DW], toxicity equivalent quotient [TEQ]) preliminary remediation goals in residential soils, and Washington Department of Ecology (11 pg/g DW TEQ) for residential soil clean-up levels (USEPA 2010). They are also generally an order of magnitude below USEPA effects-based ecological receptors thresholds (60 to 100 pg/g DW TEQ for fish; 2.5 to 25 pg/g DW TEQ for mammals; 21 to 210 pg/g DW TEQ for birds).

While the existing sediment data (Shannon & Wilson, Inc. 2006) did not indicate a high risk of sediment toxicity, it was not sufficient to evaluate all analytes of interest. Thus, as part of the Secretarial Determination studies, a sediment evaluation was undertaken during 2009 to 2011 to provide a more comprehensive data set to further guide decisionmakers in an evaluation of potential impacts from dam removal. The United

States Bureau of Reclamation (USBR) and USFWS plan expanded the number of sediment cores and the analyte suite examined, including chemicals likely to bioaccumulate, and included biological and elutriate tests (USBR 2010). In 2009 to 2011 evaluation, establishment of toxicity and/or bioaccumulative potential for sediment contaminants relied upon thresholds developed through regional and state efforts such as the 2009 Sediment Evaluation Framework (SEF) for the Pacific Northwest Oregon and ODEQ bioaccumulation screening level values (SLVs). Sediment cores were collected during 2009 to 2010 at 37 sites on the historical Klamath River channel (on-thalweg) and the surrounding submerged terraces or near tributary mouths along the edge of the historical Klamath River (off-thalweg), distributed throughout J.C. Boyle Reservoir (Figure 2.6-4), Copco No. 1 Reservoir (Figure 2.6-5), Iron Gate Reservoir (Figure 2.6-6), and the Klamath River Estuary (Figure 3.2-5) (USBR 2010, 2011). Twelve sites (7 on-thalweg, 5 off-thalweg) were sampled in J.C. Boyle Reservoir, with maximum core depths ranging from 0.3 feet near the middle of the reservoir to 18.7 feet near the dam. Twelve sites (7 on-thalweg, 5 off-thalweg) were sampled in Copco No. 1 Reservoir, with maximum core depths ranging from 1.2 feet on an off-thalweg site downstream of the Beaver Creek arm of the reservoir to 9.7 feet on an off-thalweg location upstream of the Beaver Creek arm of the reservoir. Thirteen sites (8 on-thalweg, 5 off-thalweg) were sampled in Iron Gate Reservoir, with maximum core depths ranging from 0.5 feet at the upstream end of the reservoir to 7.7 feet within the Jenny Creek arm of the reservoir. At each site, cores were inspected by on-site geologists to verify that the reservoir-deposited/pre-reservoir sediment contact had been reached for each core. Sediment cores were used to either create whole core composite sediment samples or interval composite/depth interval composite sediment samples for laboratory analysis of potential contaminants with samples representing both the reservoir-deposited and pre-reservoir sediments. Area composite samples were also generated from sediment cores for the Klamath River Estuary. A total of 77 sediment samples were created to analyze sediment conditions in J.C. Boyle Reservoir (26 sediment samples), Copco No. 1 Reservoir (25 sediment samples), Iron Gate Reservoir (24 sediment samples), and the Klamath River Estuary (2 sediment samples) from the 37 sediment cores (USBR 2011).

A total of 501 analytes were quantified across the samples, including metals, poly-aromatic hydrocarbons (PAHs), PCBs, pesticides/herbicides, phthalates, VOCs, SVOCs, dioxins, furans, and polybrominated diphenyl ethers (PBDEs) (i.e., flame retardants). The chemical composition of sediment and elutriate[5] sediment samples were analyzed, and bioassays were conducted on the sediment and elutriate sediment samples using fish and invertebrate national benchmark toxicity species. Using results of these analyses, the following five exposure pathways were evaluated under Level 2A and 2B of the SEF using multiple lines of evidence: (CDM 2011):

- Pathway 1 – Proposed Project - Short-term water column exposure for aquatic biota from sediments flushed downstream (suspended sediments, not a bioaccumulation issue).

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[5] Elutriate sediment samples are created from reservoir composite sediment samples mixed with reservoir water (e.g., one part sediment to four parts water). In general, elutriate tests are a standard approach that analyze the chemical composition of the overlying water of the elutriate sediment samples to estimate potential chemical concentrations in the water between the grains of sediment (pore water). Standard elutriate tests do not reflect the full dilution of re-suspended sediments that would occur during dam removal.

- Pathway 2 – Proposed Project - Long-term sediment exposure for riparian biota and humans from reservoir terrace deposits and river bank deposits (terrestrial exposures).

- Pathway 3 – Proposed Project - Long-term sediment exposure for aquatic biota and humans from river bed deposits (aquatic exposures).

- Pathway 4 – Proposed Project - Long-term sediment exposure for aquatic biota from estuary and marine near shore deposits.

- Pathway 5 – No Project Alternative - Long-term sediment exposure for aquatic biota and humans (via fish consumption) to reservoir sediments.

Results indicate that sediment in all three reservoirs exceeded freshwater ecological screening levels (SLs) for nickel, iron, and 2,3,4,7,8-Pentachlorodibenzofuran (PECDF) (Table C-6). Sediment in J.C. Boyle Reservoir also exceeded freshwater ecological SLs for 4,4'-Dichlorodiphenyltrichloroethane (DDT), 4,4'-Dichlorodiphenyldichloroethane (DDD), 4,4'-dichlorodiphenyldichloroethylene (DDE), dieldrin, and 2,3,7,8-Tetrachlorodibenzodioxin (TCDD) (Table C-6). Several pesticides and semi-volatile organic compounds (SVOCs) were not detected in the reservoir sediments, but the reporting limits were above the freshwater SLs. Human health SLs were only exceeded for arsenic and nickel, pentachlorophenol (in the case of J.C. Boyle Reservoir), and some legacy pesticides (e.g., 4,4'-DDT, 4,4'-DDD, 4,4'-DDE, dieldrin, see Table C-7). Several dioxin-like compounds were detected and exceeded the ODEQ Bioaccumulation SLVs (Table C-7). Several pesticides and SVOCs were not detected, but the reporting limits were above the human health SLs (Table C-7). Marine ecological SLs were only exceeded for dieldrin and 2,3,4,7,8-PECDF in J.C. Boyle Reservoir (see Table C-8). Several organic compounds were not detected, but the reporting limits were above the available marine SLs (Table C-8). Analytes that were not detected but had reporting limits above freshwater or marine SLs were listed as chemicals of potential concern (COPCs) and analyzed in macroinvertebrate and/or fish tissue bioaccumulation tests to evaluate any effects from those chemicals potentially being in the sediment. No consistent pattern of elevated chemical composition was observed across discrete sampling locations within a reservoir, but sediment in J.C. Boyle Reservoir does have marginally higher iron concentrations and more detected COPCs as compared to Copco No. 1 and Iron Gate Reservoir and Klamath River Estuary sediments (CDM 2011). Also, J.C. Boyle reservoir has more COPCs based on comparison to CalEPA, National Oceanic and Atmospheric Administration (NOAA), U.S. Fish and Wildlife Service (USFWS), USEPA, and ODEQ freshwater ecological and human health SLs.

Several chemicals identified as COPCs may occur in reservoir sediments at concentrations similar to background levels, however, background concentrations of most chemicals associated with Lower Klamath Project reservoir sediments are generally unavailable for the Klamath Basin. Arsenic concentrations in measured reservoir sediments (4.3 to 15 mg/kg) were within the range of arsenic concentrations measured in soil samples from the Mid- and Lower Klamath Basin (0.8 to 23 mg/kg with typical arsenic concentrations between approximately 2.0 and 7.0 mg/kg) (USGS NGS 2008) and arsenic may be naturally elevated in the Upper Klamath Basin, with average regional background arsenic concentrations of 3.99 mg/kg ± 5.03 mg/kg in the vicinity of the Upper Klamath Lake (Sturdevant 2010; ODEQ 2013; Sullivan and Round 2016). Suitable background sites characterized by similar sediment compositions as the Lower Klamath Project reservoirs, but without the same potential chemical sources (e.g., urban

areas, irrigated agriculture, industry, and hydroelectric development), could not be identified (CDM 2011).

Overall, there were relatively few chemicals in sediment from the three reservoirs (and the Klamath River Estuary) identified as COPCs, with several COPCs being listed for the reservoirs only because the reporting limits were greater than freshwater or marine SLs. Analytical testing methods for chemicals were selected to achieve a reporting limit 3 to 5 times lower than the lowest applicable sediment screening level or water quality criteria. Despite this, due to limitations of the standard methods, several reporting limits were greater than the associated SLs and other reporting limits were not achieved due to background effects of the sample matrix or required sample dilutions (USBR 2011). However, as part of multiple lines of evidence (see *Contaminants in Aquatic Biota* below), CDM (2011) concluded that the sediment quality of reservoirs does not appear to be notably contaminated based on the detected COPCs in reservoir sediments and comparisons to ecological freshwater and marine SEF, USACE Dredge Material Management Program (DMMP), NOAA Screening Quick Reference Tables (SQuiRT), and ODEQ bioaccumulative SLs along with comparisons to human health SLs, including USEPA residential (total carcinogenic and total non-carcinogenic) regional screening levels (RSLs), CalEPA OEHHA California Human Health Screening Levels (CHHSLs), or ODEQ bioaccumulation (human subsistence and human general) SLs.

Updates to SEF Screening Levels
In 2015, new SEF SLs were implemented by the RSET to replace the 2009 SEF SLs. RSET undertook a range of updates to the various SEF SLs, including increasing SLs, decreasing SLs, adding SLs for new chemicals, and removing SLs for chemicals listed in the 2009 SEF SLs (RSET 2009, 2018). The majority of the changes to the SLs were updates to the freshwater SEF SLs with only a few updates to the marine SEF SLs (e.g., the marine SL for DDT was revised from 34 ug/kg in 2009 to 12 ug/kg in 2018). The 2009 Pacific Northwest SEF freshwater SLs and the updated freshwater SLs detailed in the 2018 Pacific Northwest SEF are presented in Table C-9. While the value of SLs for some analytes changed between the 2009 and 2018 SEF, the changes in SLs did not alter the detected COPC list determined by CDM (2011) or the resulting analysis, since the detected COPCs were not listed only in reference to the SEF SLs. More specifically, a decrease in the lower SEF SL (SL1) for nickel between the 2009 and 2018 SEF did not change its classification in the COPC list, because nickel had already been identified as a COPC in the CDM (2011) analysis for exceedances of the FWS TEL, FWS LEL, and FWS TEC screening values. Nickel concentrations measured in reservoir sediments (18 to 33 mg/kg) were occasionally above the nickel 2018 SEF lower SL (26 mg/kg), so exceedances of the 2018 SEF lower SL would be added to the list of reasons why nickel is listed as a COPC for the reservoir sediment. While the chromium SEF SLs decreased, chromium concentrations in reservoir sediments were still below chromium 2018 SEF SLs, so it is still not a COPC for reservoir sediments. Selenium was added to the list of 2018 SEF SLs, but selenium concentrations measured in reservoir sediments were less than the 2018 SEF SLs and it would not be a COPC.

As some SEF SLs are less than the laboratory reporting limit, the COPCs listed in CDM (2011) would change with application of the 2018 SEF SLs for the following reasons:
1. a SL no longer applied for a chemical in the 2018 SEF SL;
2. a SL was added for a new chemical and the USBR (2011) reporting limit for that chemical was less than the 2018 SEF SL; or

3. a SL was added for a new chemical and the chemical had not been tested in the USBR (2011) analysis of reservoir sediments.

Analyte-specific details are provided below.

The 2009 SEF SLs for PAHs no longer appear in the 2018 SEF SLs, so they would be removed from the COPC list. Total PAHs would be added to the COPC list, because it is a new analyte in the 2018 SEF SLs and the total PAHs were not measured as part of the USBR (2011) tests. While individual PAHs were measured in reservoir sediment testing and the range of the reporting limits for individual PAHs (6.7 to 1,200 ug/kg) were below the total PAHs 2018 SEF lower SL (17,000 ug/kg), total PAHs were not measured and would need added to the COPC list. Similarly, individual aroclors (e.g., aroclor 1221) would be removed from the COPC list and total aroclors would be added to the COPC list since it was not measured as part of the USBR (2011) testing.

The 2018 SEF added SLs for several organochlorine pesticides (i.e., 4-4'-DDD, 4-4'-DDE, 4-4'-DDT, dieldrin, beta-hexachlorocyclohexane, and endrin ketone). UBSR (2011) testing did not detect any of these chemicals in reservoir sediments and the reporting limits for these chemicals were above the 2018 SEF SL, so 4-4'-DDD, 4-4'-DDE, 4-4'-DDT, dieldrin, beta-hexachlorocyclohexane, and endrin ketone would not need to be added to the COPC list based on 2018 SEF SLs.

The 2018 SEF revised the SLs for phthalates (Bis(2-ethylhexyl) phthalate and Di-N-octyl phthalate), removed SLs for phthalates (Butyl benzyl phthalate and Dimethyl phthalate), and added SLs for phthalates (Di-n-butyl-phthalate). None of the phthalates were detected in reservoir sediments above the reporting limit, but all the phthalates with 2018 SEF SLs would be retained or added to the COPC list since their reporting limit was below the 2018 SEF SLs. The phthalates without 2018 SEF SLs (Butyl benzyl phthalate and Dimethyl phthalate) would be removed from the COPC list.

The 2018 SEF added SLs for several SVOC phenols (Phenol, 4-Methylphenol, and Pentachlorophenol). Pentachlorophenol was not detected in USBR (2011) reservoir sediments and the reporting limit was below the 2018 SEF SLs, so it would not need to be added to the COPC list. Phenol and 4-Methylphenol were not detected in reservoir sediments, but the reporting limit was greater than the 2018 SEF SLs, so it would be added to the COPC list.

SLs were also added in the 2018 SEF for the extractable compounds benzoic acid, carbazole, and dibenzofuran. These three chemicals were not detected in USBR (2011) sediment tests, but the range of reporting limits was greater than the 2018 SEF SLs. As such, they would be added to the COPCs list.

Several new chemicals were added to the list of chemicals with 2018 SEF SLs that had not been previously considered or measured in USBR (2011) reservoir sediments tests. Butyltins are a group of organotin compounds used as stabilizers for polyvinyl chloride (PVC), biocides, fungicides, and anti-biofouling agents, especially used in coatings and paints applied to the bottom hull of ships to minimize biofouling (USEPA 2003; RSET 2018). Tributyltin is the primarily form of butyltin of concern in the environment due to its use as an anti-biofouling agent in paints for large ships with other forms (mono-, di-, and tetra-butyltin) occurring as byproducts as tributyltin breaks down. Elevated concentrations of butyltins in fresh and salt waters, sediments, and biota are primarily associated with harbors, marinas, boat yards, and dry docks with frequent ship traffic,

but use of butyltin as an anti-biofouling agent on nets, crab pots, docks, and water cooling towers also contributes to its presence in aquatic environments (WHO 1999; USEPA 2003). According to RSET (2018), the need for analysis of butyltins (and other organotins) is limited to areas and sites affected by vessel maintenance and construction activities, marine shipping, and frequent vessel traffic (e.g., shipyards, boatyards, marinas, and marine terminals). Conditions in the Hydroelectric Reach of the Klamath River and upstream reaches did not support these activities historically and currently the reservoirs do not experience frequent large vessel traffic, therefore the site-specific SEF butyltin SLs would not apply to the Lower Klamath Project.

Total Petroleum Hydrocarbons (TPHs) represent gasoline, diesel, and other petroleum hydrocarbon compounds (e.g., motor oil or grease). While many other individual hydrocarbons that are present in gasoline, diesel, and petroleum products (e.g., PAHs, VOCs, SVOCs) were measured by USBR (2011), the overarching TPH test was not performed, thus TPHs would need to be added to the COPC list.

While the COPCs list would change due to the changes in SLs between the 2009 SEF and 2018 SEF, the previous CDM (2011) analysis of the five exposure pathways still sufficiently analyzes the potential effects of chemicals in Lower Klamath Project reservoir sediments through sediment toxicity and bioaccumulation tests for macroinvertebrates and/or fish. Since the sediment toxicity and bioaccumulation tests were performed using sediment samples from the Lower Klamath Project reservoirs, the results quantify the integrative (cumulative) effects of all COPCs in the reservoir sediments, even if the full updated list of COPCs was not included in the CDM (2011) analysis. Additionally, SEF SLs were only one of three ecological freshwater sediment SLs used in the original analysis, so the other SLs (USACE DMMP, NOAA SQuiRT, and ODEQ bioaccumulation) ensure that the CDM (2011) analysis sufficiently assessed the inorganic and organic contaminants in reservoir sediment that may impact freshwater aquatic species. Finally, SEF SLs were not used as human health screening levels, so the changes in the SEF SLs would not alter the CDM (2011) analysis with respect to reservoir sediments on human health.

Sediment Toxicity and Bioaccumulation Tests
Toxicity equivalent quotients (TEQs) were calculated for dioxin, furan, and dioxin-like PCBs in reservoir sediment samples to evaluate potential adverse effects from exposure to dioxin, furan, and dioxin-like PCBs. TEQs ranged from approximately 4 to 9 pg/g for J.C. Boyle Reservoir, 5 to 10 pg/g for Copco No. 1 Reservoir, and 2 to 4 pg/g for Iron Gate Reservoir. In some cases, these values are slightly higher than background values reported by USEPA for Region 9 (i.e., 2 to 5 pg/g), Region 10 (i.e., 4 pg/g), and for non-impacted lakes of the United States (i.e., 5.3 pg/g) (USEPA 2010, CDM 2011). The calculated TEQs may also be within the range of local background values. Since the TEQs are only slightly above regional background concentrations and the nationwide background for non-impacted lakes, they have limited potential for adverse effects for fish exposed to reservoir sediments (CDM 2011).

Toxicity tests generally indicated low potential for sediment toxicity to benchmark benthic indicator species with the exception of a single sample from J.C. Boyle Reservoir, where survival of the benthic amphipod *Hyalella azteca* indicated a moderate potential for sediment toxicity (CDM 2011). Results of elutriate chemistry and elutriate toxicity tests on rainbow trout (*Oncorhynchus mykiss*) are discussed as part of the Proposed Project potential impact analysis (Section 3.2.5.7 *Inorganic and Organic Contaminants*).

Collectively, the elutriate chemistry and elutriate toxicity test results do not identify a consistent pattern of toxicity by location, representative organism, or conditions.

Overall, twenty lines of evidence were used with various lines of evidence integrated to evaluate the five exposure pathways and to draw conclusions regarding potential adverse effects from the chemicals present in the reservoir sediments (Table C-10). Lines of evidence related to contaminants in aquatic biota are detailed and discussed together below. Based on these twenty lines of evidence from the 2009 to 2010 Secretarial Determination study, reservoir sediments do not appear to be highly contaminated (CDM 2011). No consistent pattern of elevated chemical composition is observed across discrete sampling locations within a reservoir. No single reservoir was observed to be consistently more or less contaminated based on these 20 lines of evidence. Where elevated concentrations of chemicals in sediment are found, the degree of exceedance based on comparisons of measured (i.e., detected) chemical concentrations to SLs is small, and in several cases, may reflect regional background conditions (CDM 2011).

Table C-6. Chemicals in Sediment that Exceed One or More Freshwater Sediment Screening Levels. Source: CDM (2011).

Chemical	COC Based on Detect (D) or Elevated Reporting Limit (RL)	Units	Range of Detections for Detected Analytes that Exceed One or More Screening Levels	Range of Reporting Limits (RL) for Non-Detects	Ratio of Maximum Chemical Concentration to SL for Detected Analytes[a]	Screening Values Exceeded	Highest of Screening Value Hierarchy Level[b]
J.C. Boyle Reservoir							
Nickel	D	mg/kg	19–32	---	2 [c]	FWS TEL, **FWS LEL** [c], FWS TEC	2d [c]
4,4-DDD	D	ug/kg	3.7	---	9.5	ODEQ Bioacc SLV	2c
4,4-DDE	D	ug/kg	3.4	---	8.7	ODEQ Bioacc SLV	2c
4,4-DDT	D	ug/kg	4.1	---	11	ODEQ Bioacc SLV	2c
Dieldrin	D	ug/kg	3.4	---	1.5–9.2	FWS TEL, FWS LEL, FWS TEC, **ODEQ F-FW, ODEQ B-I**, ODEQ B-P, ODEQ M-I, ODEQ M-P	2c
2,3,4,7,8-PECDF	D	pg/g	1.5–1.5	---	1.4–8.8	**ODEQ F-FW**, ODEQ B-I, **ODEQ M-I**	2c
2,3,7,8-TCDD	D	pg/g	0.19	---	3.7	ODEQ M-I	2c
Iron	D	mg/kg	21,000–37,000	---	1.85	FWS LEL	2d
Cadmium	RL	mg/kg	---	0.16–0.84	---	SEF-SL1	2a
Aroclor 1221	RL	ug/kg	---	0.24–0.49	---	SEF-SL1 (total PCBs)	2a
Aroclor 1232	RL	ug/kg	---	0.16–0.24	---	SEF-SL1 (total PCBs)	2a
Aroclor 1242	RL	ug/kg	---	0.045–0.24	---	SEF-SL1 (total PCBs)	2a
Aroclor 1248	RL	ug/kg	---	0.045–0.24	---	SEF-SL1 (total PCBs)	2a
Aroclor 1254	RL	ug/kg	---	0.045–0.24	---	SEF-SL1 (total PCBs)	2a
Aroclor 1260	RL	ug/kg	---	0.045–0.24	---	SEF-SL1 (total PCBs)	2a
Bis(2-ethylhexyl)phthalate	RL	ug/kg	---	230–1200	---	SEF-SL1	2a
Butyl benzyl phthalate	RL	ug/kg	---	230–1200	---	SEF-SL1	2a
Dimethyl phthalate	RL	ug/kg	---	230–1200	---	SEF-SL1	2a
Di-n-octyl phthalate	RL	ug/kg	---	230–1200	---	SEF-SL1	2a

Chemical	COC Based on Detect (D) or Elevated Reporting Limit (RL)	Units	Range of Detections for Detected Analytes that Exceed One or More Screening Levels	Range of Reporting Limits (RL) for Non-Detects	Ratio of Maximum Chemical Concentration to SL for Detected Analytes[a]	Screening Values Exceeded	Highest of Screening Value Hierarchy Level[b]
2-METHYLNAPHTHALENE	RL	ug/kg	---	230–1200	---	SEF-SL1	2a
ACENAPHTHENE	RL	ug/kg	---	230–1200	---	SEF-SL1	2a
ACENAPHTHYLENE	RL	ug/kg	---	230–1200	---	SEF-SL1	2a
BENZO(K)FLUORANTHENE	RL	ug/kg	---	230–1200	---	SEF-SL1	2a
DIBENZ(A,H)ANTHRACENE	RL	ug/kg	---	230–1200	---	SEF-SL1	2a
DIBENZOFURAN	RL	ug/kg	---	230–1200	---	SEF-SL1	2a
FLUORENE	RL	ug/kg	---	230–1200	---	SEF-SL1	2a
Chlordane (Technical)	RL	ug/kg	---	4.5–24	---	ODEQ Bioacc SLV	2c
Chlordane-Alpha	RL	ug/kg	---	0.9–4.9	---	ODEQ Bioacc SLV	2c
Chlordane-Gamma	RL	ug/kg	---	0.9–4.9	---	ODEQ Bioacc SLV	2c
Dieldrin	RL	ug/kg	---	0.9–4.9	---	ODEQ Bioacc SLV	2c
BHC-Gamma (HCH-gamma, Lindane)	RL	ug/kg	---	0.9–4.9	---	SQuiRTs (TEL, LEL, PEL, TEC)	2d
Endrin	RL	ug/kg	---	0.9–4.9	---	SQuiRTs (TEL, LEL, TEC)	2d
Heptachlor	RL	ug/kg	---	0.9–4.9	---	SQuiRTs (TEL, LEL, TEC)	2d
Heptachlor Epoxide	RL	ug/kg	---	0.9–4.9	---	SQuiRTs (TEL, PEL, TEC)	2d
Toxaphene	RL	ug/kg	---	45–240	---	SQuiRTs (TEL)	2d
Copco No. 1 Reservoir							
Nickel	D	mg/kg	22–32	---	2 [c]	FWS TEL, **FWS LEL** [c], FWS TEC	2d [c]
2,3,4,7,8-PECDF	D	pg/g	1.8–1.9	---	1.7–11.2	**ODEQ F-FW, ODEQ B-I, ODEQ M-I**	2c
Iron	D	mg/kg	21,000–24,000	---	1.2	FWS LEL	2d
SILVER	RL	mg/kg	---	1.8–2.4	---	SEF-SL1	2a
AROCLOR 1221	RL	ug/kg	---	0.24–0.3	---	SEF-SL1 (total PCBs)	2a
AROCLOR 1232	RL	ug/kg	---	0.12–0.15	---	SEF-SL1 (total PCBs)	2a

Chemical	COC Based on Detect (D) or Elevated Reporting Limit (RL)	Units	Range of Detections for Detected Analytes that Exceed One or More Screening Levels	Range of Reporting Limits (RL) for Non-Detects	Ratio of Maximum Chemical Concentration to SL for Detected Analytes[a]	Screening Values Exceeded	Highest of Screening Value Hierarchy Level[b]
BIS(2-ETHYLHEXYL) PHTHALATE	RL	ug/kg	---	580–730	---	SEF-SL1	2a
BUTYL BENZYL PHTHALATE	RL	ug/kg	---	580–730	---	SEF-SL1	2a
DIMETHYL PHTHALATE	RL	ug/kg	---	580–730	---	SEF-SL1	2a
DI-N-OCTYL PHTHALATE	RL	ug/kg	---	580–730	---	SEF-SL1	2a
2-METHYLNAPHTHALENE	RL	ug/kg	---	580–730	---	SEF-SL1	2a
ACENAPHTHYLENE	RL	ug/kg	---	580–730	---	SEF-SL1	2a
BENZO(K)FLUORANTHENE	RL	ug/kg	---	580–730	---	SEF-SL1	2a
DIBENZOFURAN	RL	ug/kg	---	580–730	---	SEF-SL1	2a
4,4'-DDE	RL	ug/kg	---	2.4–3	---	SQuiRTs (TEL)	2d
4,4'-DDT	RL	ug/kg	---	2.4–3	---	SQuiRTs (TEL)	2d
BHC-gamma (HCH-gamma, Lindane)	RL	ug/kg	---	2.4–3	---	SQuiRTs (TEL, PEL, TEC)	2d
CHLORDANE (TECHNICAL)	RL	ug/kg	---	12–15	---	SQuiRTs (TEL, LEL, PEL, TEC)	2d
DIELDRIN	RL	ug/kg	---	2.4–3	---	SQuiRTs (TEL, LEL, TEC)	2d
ENDRIN	RL	ug/kg	---	2.4–3	---	SQuiRTs (TEC)	2d
HEPTACHLOR	RL	ug/kg	---	2.4–3	---	SQuiRTs (TEL, TEC)	2d
HEPTACHLOR EPOXIDE	RL	ug/kg	---	2.4–3	---	SQuiRTs (TEL, PEL, TEC)	2d
TOXAPHENE	RL	ug/kg	---	120–150	---	SQuiRTs (TEL)	2d

Chemical	COC Based on Detect (D) or Elevated Reporting Limit (RL)	Units	Range of Detections for Detected Analytes that Exceed One or More Screening Levels	Range of Reporting Limits (RL) for Non-Detects	Ratio of Maximum Chemical Concentration to SL for Detected Analytes[a]	Screening Values Exceeded	Highest of Screening Value Hierarchy Level[b]
Iron Gate Reservoir							
Nickel	D	mg/kg	18–33	---	2.1 [c]	FWS TEL, **FWS LEL** [c], FWS TEC	2d [c]
2,3,4,7,8-PECDF	D	pg/g	0.74	---	1.1–4.4	ODEQ B-I, ODEQ M-I	2c
Iron	D	mg/kg	26,000–32,000	---	1.6	FWS LEL	2d
SILVER	RL	mg/kg	---	0.94–2.2	---	SEF-SL1	2a
AROCLOR 1221	RL	ug/kg	---	0.067–0.3	---	SEF-SL1 (total PCBs)	2a
AROCLOR 1232	RL	ug/kg	---	0.033–0.15	---	SEF-SL1 (total PCBs)	2a
BIS(2-ETHYLHEXYL) PHTHALATE	RL	ug/kg	---	170–730	---	SEF-SL1	2a
BUTYL BENZYL PHTHALATE	RL	ug/kg	---	170–730	---	SEF-SL1	2a
DI-N-OCTYL PHTHALATE	RL	ug/kg	---	170–730	---	SEF-SL1	2a
2-METHYLNAPHTHALENE	RL	ug/kg	---	170–730	---	SEF-SL1	2a
ACENAPHTHYLENE	RL	ug/kg	---	170–730	---	SEF-SL1	2a
BENZO(K)FLUORANTHENE	RL	ug/kg	---	170–730	---	SEF-SL1	2a
DIBENZOFURAN	RL	ug/kg	---	170–730	---	SEF-SL1	2a
NAPHTHALENE	RL	ug/kg	---	5–520	---	SEF-SL1	2a
4,4'-DDE	RL	ug/kg	---	0.67–3	---	SQuiRTs (TEL, TEC)	2d
4,4'-DDT	RL	ug/kg	---	0.67–3	---	SQuiRTs (TEL)	2d
BHC-gamma (HCH-gamma, Lindane)	RL	ug/kg	---	0.67–3	---	SQuiRTs (TEL, PEL, TEC)	2d
CHLORDANE (TECHNICAL)	RL	ug/kg	---	3.3–15	---	SQuiRTs (TEL, PEL, LEL, TEC)	2d
DIELDRIN	RL	ug/kg	---	0.67–3	---	SQuiRTs (TEL, LEL, TEC)	2d
ENDRIN	RL	ug/kg	---	0.67–3	---	SQuiRTs (TEL, TEC)	2d
HEPTACHLOR	RL	ug/kg	---	0.67–3	---	SQuiRTs (TEL, TEC)	2d
HEPTACHLOR EPOXIDE	RL	ug/kg	---	0.67–3	---	SQuiRTs (TEL, PEL, TEC)	2d
TOXAPHENE	RL	ug/kg	---	33–150	---	SQuiRTs (TEL)	2d

Chemical	COC Based on Detect (D) or Elevated Reporting Limit (RL)	Units	Range of Detections for Detected Analytes that Exceed One or More Screening Levels	Range of Reporting Limits (RL) for Non-Detects	Ratio of Maximum Chemical Concentration to SL for Detected Analytes[a]	Screening Values Exceeded	Highest of Screening Value Hierarchy Level[b]
Klamath River Estuary							
Lower Klamath							
Chromium	D	mg/kg	96	--	1.0	**SL1-FWS**, SL2-FWS, FWS TEL, FWS LEL, FWS PEL, FWS TEC	2b
Nickel	D	mg/kg	110	--	1.8 [c]	**SL1-FWS,** SL2-FWS, FWS TEL, FWS LEL, FWS PEL, FWS TEC	2b
Iron	D	mg/kg	24,000–24,000	--	1.2	**FWS LEL**	2d
BIS(2-ETHYLHEXYL) PHTHALATE	RL	ug/kg	---	230	---	**SEF-SL1**	2a
DIMETHYL PHTHALATE	RL	ug/kg	---	230	---	**SEF-SL1**	2a
DI-N-OCTYL PHTHALATE	RL	ug/kg	---	230	---	**SEF-SL1**	2a
CHLORDANE (TECHNICAL)	RL	ug/kg	---	4.6	---	**SQuiRTs (TEL, TEC)**	2d
HEPTACHLOR EPOXIDE	RL	ug/kg	---	0.91	---	**SQuiRTs (TEL)**	2d
TOXAPHENE	RL	ug/kg	---	46	---	**SQuiRTs (TEL)**	2d
Upper Klamath							
Chromium	D	mg/kg	96–97	---	1.0	**SL1-FWS,** FWS TEL, FWS LEL, FWS PEL, FWS TEC	2a
Bis(2-ethylhexyl)phthalate	D	ug/kg	250	---	1.1	**SL1-FWS**	2a
DIMETHYL PHTHALATE	RL	ug/kg	---	230	---	**SEF-SL1**	2a
DI-N-OCTYL PHTHALATE	RL	ug/kg	---	230	---	**SEF-SL1**	2a
CHLORDANE (TECHNICAL)	RL	ug/kg	---	4.6	---	**SQuiRTs (TEL, TEC)**	2d

Chemical	COC Based on Detect (D) or Elevated Reporting Limit (RL)	Units	Range of Detections for Detected Analytes that Exceed One or More Screening Levels	Range of Reporting Limits (RL) for Non-Detects	Ratio of Maximum Chemical Concentration to SL for Detected Analytes[a]	Screening Values Exceeded	Highest of Screening Value Hierarchy Level[b]
HEPTACHLOR EPOXIDE	RL	ug/kg	---	0.93	---	**SQuiRTs (TEL)**	2d
TOXAPHENE	RL	ug/kg	---	46	---	**SQuiRTs (TEL)**	2d

Notes:

Screening Level Hierarchy --
Retain if above:

1) DMMP-MLs

2a) SEF-SL1
2b) SEF-SL1 AND SEF-SL2
2c) Chemicals with no SEF and one or more ODEQ bioaccumulative SLVs exceeded
2d) Chemicals with no SEF or ODEQ but one or more SQuiRT exceeded

Units:

Metals:	mg/kg
Pesticides:	ug/kg
dioxins and furans:	pg/g
SVOCs:	ug/kg
phthalates:	ug/kg

Key:

FWS =	U.S. Fish and Wildlife Service	DMMP =	Dredged Material Management Program	TEL =	threshold effect level
TEL =	Threshold Effect Level	LEL =	Lowest Effect level	LEL =	lowest effect level
SL1 =	Sediment Screening Level 1	TEC =	Threshold Effect Concentration	PEL =	probable effect level
SEF =	Sediment Evaluation Framework	PEL =	Probable Effect Level	TEC =	threshold effect concentration
SLV =	Screening Level Value	ODEQ:	Oregon Department of Environmental Quality		B-P: Bird population
	B-I: Bird individual		M-I: Mammal individual		M-P: Mammal population
	F-FW: Fish-freshwater				

[a] Ratio of maximum detected concentration to the SL is typically expressed as a Hazard Quotient (HQ). This ratio is presented above for each detected chemical and is calculated using the maximum detected concentration; the highest and lowest of screening values when multiple screening values are exceeded of same level in screening hierarchy. When more than two screening values are exceeded, the screening level used for calculation of the ratio (HQ) are in **bold**.

[b] Screening level hierarchy depicted in CDM (2011) Figure 2.

[c] Updated from CDM (2011) Table 2 based on review of screening values listed in CDM (2011) Table A-2 and range of detections for detected analytes in CDM (2011) and USBR (2011).

Based on the information provided in Table A-6 in CDM (2011) and database query for ambiguous and positive exceedances.

Table C-7. Chemicals in Sediment that Exceed One or More Human Health Sediment Screening Levels. Source: CDM (2011).

Chemical	COC Based on Detect (D) or Elevated Reporting Limit (RL)	Units	Range of Detections for Detected Analytes that Exceed One or More Screening Levels	Range of Reporting Limits (RL) for Non-Detects	Ratio of Maximum Chemical Concentration to SL for Detected Analytes[1]	Screening Values Exceeded	Notes
J.C. Boyle Reservoir							
Arsenic	D	mg/kg	4.3–15	---	38–214	**EPA RSL TOT CAR**, **CHHSL Res**, CHHSL Comm	a
Nickel	D	mg/kg	19–32	---	84	**EPA RSL TOT CAR**	a
4,4-DDD	D	ug/kg	3.7	---	11–93	ODEQ BSLV H-S, ODEQ BSLV H-G	---
4,4-DDE	D	ug/kg	3.4	---	10–85	ODEQ BSLV H-S, ODEQ BSLV H-G	---
4,4-DDT	D	ug/kg	4.1	---	12–103	ODEQ BSLV H-S, ODEQ BSLV H-G	---
Dieldrin	D	ug/kg	3.4	---	420–3,400	ODEQ BSLV H-S, ODEQ BSLV H-G	---
1,2,3,4,6,7,8-HPCDD	D	pg/g	170–180	---	2.1	ODEQ BSLV H-S	---
1,2,3,4,7,8-HXCDD	D	pg/g	1.5–1.6	---	4.4	ODEQ BSLV H-S	---
1,2,3,6,7,8-HXCDD	D	pg/g	6.6–7.3	---	2.7–21	ODEQ BSLV H-S, ODEQ BSLV H-G	---
1,2,3,7,8,9-HXCDD	D	pg/g	3.7	---	1.4–11	ODEQ BSLV H-S, ODEQ BSLV H-G	---
1,2,3,4,7,8-HXCDF	D	pg/g	1.7–2.1	---	6.2	ODEQ BSLV H-S	---
1,2,3,6,7,8-HXCDF	D	pg/g	4.4–5.3	---	2.0–16	ODEQ BSLV H-S, ODEQ BSLV H-G	---
1,2,3,7,8,9-HXCDF	D	pg/g	0.66–0.67	---	0.5–1.9	ODEQ BSLV H-S	---
1,2,3,7,8-PECDD	D	pg/g	1.1	---	4.1–37	ODEQ BSLV H-S, ODEQ BSLV H-G	---
1,2,3,7,8-PECDF	D	pg/g	0.88–1.1	---	3.5	ODEQ BSLV H-S	---
2,3,4,6,7,8-HXCDF	D	pg/g	3–3.2	---	1.2–9.4	ODEQ BSLV H-S, ODEQ BSLV H-G	---
2,3,4,7,8-PECDF	D	pg/g	1.5	---	50–405	ODEQ BSLV H-S, ODEQ BSLV H-G	---
2,3,7,8-TCDD	D	pg/g	0.19	---	19–173	ODEQ BSLV H-S, ODEQ BSLV H-G	b
2,3,7,8-TCDF	D	pg/g	0.88–0.9	---	1.2–9.6	ODEQ BSLV H-S, ODEQ BSLV H-G	---
Pentachlorophenol	D	ug/kg	34	---	1.1	ODEQ BSLV H-S	b
4,4,'-DDD	RL	ug/kg	---	0.9–4.9	---	ODEQ	---
4,4,'-DDE	RL	ug/kg	---	0.9–4.9	---	ODEQ	---
4,4,'-DDT	RL	ug/kg	---	0.9–4.9	---	ODEQ	---

Chemical	COC Based on Detect (D) or Elevated Reporting Limit (RL)	Units	Range of Detections for Detected Analytes that Exceed One or More Screening Levels	Range of Reporting Limits (RL) for Non-Detects	Ratio of Maximum Chemical Concentration to SL for Detected Analytes[1]	Screening Values Exceeded	Notes
Aroclor 1221	RL	ug/kg	---	0.24–0.49	---	EPA RSL	---
Aroclor 1232	RL	ug/kg	---	0.16–0.24	---	EPA RSL	---
Aroclor 1242	RL	ug/kg	---	0.045–0.24	---	EPA RSL	---
Aroclor 1248	RL	ug/kg	---	0.045–0.24	---	EPA RSL	---
Aroclor 1254	RL	ug/kg	---	0.045–0.24	---	EPA RSL	---
Aroclor 1260	RL	ug/kg	---	0.045–0.24	---	EPA RSL	---
BHC-Gamma (HCH-gamma, Lindane)	RL	ug/kg	---	0.9–4.9	---	CHHSLs	---
Chlordane (Technical)	RL	ug/kg	---	4.5–24	---	ODEQ	---
Chlordane-Alpha	RL	ug/kg	---	0.9–4.9	---	ODEQ	---
Chlordane-Gamma	RL	ug/kg	---	0.9–4.9	---	ODEQ	---
1,2,3-TRICHLOROPROPANE	RL	ug/kg	---	6.7–36	---	EPA RSL	---
1,2-DIBROMO-3-CHLOROPROPANE	RL	ug/kg	---	6.7–36	---	EPA RSL	---
3,3'-DICHLOROBENZIDINE	RL	ug/kg	---	230–1200	---	EPA RSL	---
BENZ(A)ANTHRACENE	RL	ug/kg	---	230–1200	---	EPA RSL	---
BENZO(A)PYRENE	RL	ug/kg	---	230–1200	---	EPA RSL, CHHSLs	---
BENZO(B)FLUORANTHENE	RL	ug/kg	---	230–1200	---	EPA RSL	---
BIS(2-CHLOROETHYL) ETHER	RL	ug/kg	---	230–1200	---	EPA RSL	---
DIBENZ(A,H)ANTHRACENE	RL	ug/kg	---	230–1200	---	EPA RSL	---
FLUORENE	RL	ug/kg	---	230–1200	---	EPA RSL	---
HEXACHLOROBENZENE	RL	ug/kg	---	230–1200	---	ODEQ, USEPA RSL	---
INDENO(1,2,3-CD)PYRENE	RL	ug/kg	---	230–1200	---	EPA RSL	---
N-NITROSODI-N-PROPYLAMINE	RL	ug/kg	---	230–1200	---	EPA RSL	---

Chemical	COC Based on Detect (D) or Elevated Reporting Limit (RL)	Units	Range of Detections for Detected Analytes that Exceed One or More Screening Levels	Range of Reporting Limits (RL) for Non-Detects	Ratio of Maximum Chemical Concentration to SL for Detected Analytes[1]	Screening Values Exceeded	Notes
TRANS-1,4-DICHLORO-2-BUTENE	RL	ug/kg	---	6.7–36	---	EPA RSL	---
1,2-DIBROMOETHANE	RL	ug/kg	---	6.7–36	---	EPA RSL	---
Copco No. 1 Reservoir							
Arsenic	D	mg/kg	6.3–13	---	33–186	**EPA RSL TOT CAR, CHHSL Res, CHHSL Comm**	---
Nickel	D	mg/kg	22–32	---	84	EPA RSL TOT CAR	---
1,2,3,4,6,7,8-HPCDD	D	pg/g	180–190	---	---	----	c
1,2,3,4,6,7,8-HPCDF	D	pg/g	89–96	---	---	----	c
1,2,3,4,7,8-HXCDD	D	pg/g	1.7–1.9	---	---	----	c
1,2,3,6,7,8-HXCDD	D	pg/g	8.8–9.8	---	---	----	c
1,2,3,7,8,9-HXCDD	D	pg/g	4.2–4.3	---	---	----	c
1,2,3,4,7,8-HXCDF	D	pg/g	2.3–2.8	---	---	----	c
1,2,3,6,7,8-HXCDF	D	pg/g	3.5–5.5	---	---	----	c
1,2,3,7,8,9-HXCDF	D	pg/g	1.0	---	---	----	c
2,3,4,6,7,8-HXCDF	D	pg/g	3.2–3.7	---	---	----	c
1,2,3,7,8-PECDD	D	pg/g	1.2–1.4	---	---	----	c
1,2,3,7,8-PECDF	D	pg/g	0.84	---	---	----	c
2,3,4,7,8-PECDF	D	pg/g	1.8–1.9	---	---	----	c
2,3,7,8-TCDF	D	pg/g	0.99–1.2	---	---	---	c
AROCLOR 1221	RL	ug/kg	---	0.24–0.3	---	EPA RSL	---
AROCLOR 1232	RL	ug/kg	---	0.12–0.15	---	EPA RSL	---
BHC-gamma (HCH-gamma, Lindane)	RL	ug/kg	---	2.4–3	---	CHHSLs	---
1,2,3-TRICHLOROPROPANE	RL	ug/kg	---	18–22	---	EPA RSL	---

Chemical	COC Based on Detect (D) or Elevated Reporting Limit (RL)	Units	Range of Detections for Detected Analytes that Exceed One or More Screening Levels	Range of Reporting Limits (RL) for Non-Detects	Ratio of Maximum Chemical Concentration to SL for Detected Analytes[1]	Screening Values Exceeded	Notes
1,2-DIBROMO-3-CHLOROPROPANE	RL	ug/kg	---	18–22	---	EPA RSL	---
BENZ(A)ANTHRACENE	RL	ug/kg	---	580–730	---	EPA RSL	---
BENZO(A)PYRENE	RL	ug/kg	---	580–730	---	EPA RSL, CHHSLs	---
BENZO(B)FLUORANTHENE	RL	ug/kg	---	580–730	---	EPA RSL	---
BIS(2-CHLOROETHYL) ETHER	RL	ug/kg	---	580–730	---	EPA RSL	---
DIBENZ(A,H)ANTHRACENE	RL	ug/kg	---	580–730	---	EPA RSL	---
HEXACHLOROBENZENE	RL	ug/kg	---	580–730	---	EPA RSL	---
INDENO(1,2,3-CD)PYRENE	RL	ug/kg	---	580–730	---	EPA RSL	---
N-NITROSODI-N-PROPYLAMINE	RL	ug/kg	---	580–730	---	EPA RSL	---
TRANS-1,4-DICHLORO-2-BUTENE	RL	ug/kg	---	18–22	---	EPA RSL	---
Iron Gate Reservoir							
Arsenic	D	mg/kg	7.4–10	---	26–143	**EPA RSL TOT CAR, CHHSL Res, CHHSL Comm**	---
Nickel	D	mg/kg	18–33	---	87	EPA RSL TOT CAR	---
1,2,3,4,7,8-HXCDD	D	pg/g	1.1	---	---	----	c
1,2,3,6,7,8-HXCDD	D	pg/g	3.4–3.5	---	---	----	c
1,2,3,7,8,9-HXCDD	D	pg/g	2–2.5	---	---	----	c
1,2,3,4,7,8-HXCDF	D	pg/g	1.2	---	---	----	c
1,2,3,6,7,8-HXCDF	D	pg/g	1.2–1.4	---	---	----	c
2,3,4,6,7,8-HXCDF	D	pg/g	1.2–1.4	---	---	----	c
1,2,3,7,8-PECDD	D	pg/g	0.62–0.82	---	---	----	c
1,2,3,7,8-PECDF	D	pg/g	0.44–0.52	---	---	----	c

Chemical	COC Based on Detect (D) or Elevated Reporting Limit (RL)	Units	Range of Detections for Detected Analytes that Exceed One or More Screening Levels	Range of Reporting Limits (RL) for Non-Detects	Ratio of Maximum Chemical Concentration to SL for Detected Analytes[1]	Screening Values Exceeded	Notes
2,3,4,7,8-PECDF	D	pg/g	0.74	---	---	----	c
2,3,7,8-TCDF	D	pg/g	0.68	---	---	---	c
AROCLOR 1221	RL	ug/kg	---	0.067–0.3	---	EPA RSL	---
AROCLOR 1232	RL	ug/kg	---	0.033–0.15	---	EPA RSL	---
1,2,3-TRICHLOROPROPANE	RL	ug/kg	---	5–22	---	EPA RSL	---
1,2-DIBROMO-3-CHLOROPROPANE	RL	ug/kg	---	5–22	---	EPA RSL	---
BENZ(A)ANTHRACENE	RL	ug/kg	---	170–730	---	EPA RSL	---
BENZO(A)PYRENE	RL	ug/kg	---	170–730	---	EPA RSL, CHHSLs	---
BENZO(B)FLUORANTHENE	RL	ug/kg	---	170–730	---	EPA RSL	---
BIS(2-CHLOROETHYL) ETHER	RL	ug/kg	---	170–730	---	EPA RSL	---
DIBENZ(A,H)ANTHRACENE	RL	ug/kg	---	170–730	---	EPA RSL	---
HEXACHLOROBENZENE	RL	ug/kg	---	170–730	---	EPA RSL	---
INDENO(1,2,3-CD)PYRENE	RL	ug/kg	---	170–730	---	EPA RSL	---
N-NITROSODI-N-PROPYLAMINE	RL	ug/kg	---	170–730	---	EPA RSL	---
TRANS-1,4-DICHLORO-2-BUTENE	RL	ug/kg	---	5–22	---	EPA RSL	---
Klamath River Estuary							
Lower Klamath River Estuary							
Arsenic	D	mg/kg	3.2	---	8.2–46	**EPA RSL TOT CAR, CHHSL Res,** CHHSL Comm	---
Nickel	D	mg/kg	110	---	289	EPA RSL TOT CAR	---
BHC-gamma (HCH-gamma, Lindane)	RL	ug/kg	---	0.91	---	CHHSLs	---

Chemical	COC Based on Detect (D) or Elevated Reporting Limit (RL)	Units	Range of Detections for Detected Analytes that Exceed One or More Screening Levels	Range of Reporting Limits (RL) for Non-Detects	Ratio of Maximum Chemical Concentration to SL for Detected Analytes[1]	Screening Values Exceeded	Notes
1,2,3-TRICHLOROPROPANE	RL	ug/kg	---	6.8	---	EPA RSL	---
1,2-DIBROMO-3-CHLOROPROPANE	RL	ug/kg	---	6.8	---	EPA RSL	---
BENZ(A)ANTHRACENE	RL	ug/kg	---	230	---	EPA RSL	---
BENZO(A)PYRENE	RL	ug/kg	---	230	---	EPA RSL, CHHSLs	---
BENZO(B)FLUORANTHENE	RL	ug/kg	---	230	---	EPA RSL	---
BIS(2-CHLOROETHYL) ETHER	RL	ug/kg	---	230	---	EPA RSL	---
DIBENZ(A,H)ANTHRACENE	RL	ug/kg	---	230	---	EPA RSL	---
INDENO(1,2,3-CD)PYRENE	RL	ug/kg	---	230	---	EPA RSL	---
N-NITROSODI-N-PROPYLAMINE	RL	ug/kg	---	230	---	EPA RSL	---
Upper Klamath River Estuary							
Arsenic	D	mg/kg	2.2	---	5.6–31	**EPA RSL TOT CAR, CHHSL Res,** CHHSL Comm	---
Nickel	D	mg/kg	110	---	289	EPA RSL TOT CAR	---
BHC-gamma (HCH-gamma, Lindane)	RL	ug/kg	---	0.93	---	CHHSLs	---
1,2,3-TRICHLOROPROPANE	RL	ug/kg	---	7	---	EPA RSL	---
1,2-DIBROMO-3-CHLOROPROPANE	RL	ug/kg	---	7	---	EPA RSL	---
BENZ(A)ANTHRACENE	RL	ug/kg	---	230	---	EPA RSL	---
BENZO(A)PYRENE	RL	ug/kg	---	230	---	EPA RSL, CHHSLs	---
BENZO(B)FLUORANTHENE	RL	ug/kg	---	230	---	EPA RSL	---
BIS(2-CHLOROETHYL) ETHER	RL	ug/kg	---	230	---	EPA RSL	---

Chemical	COC Based on Detect (D) or Elevated Reporting Limit (RL)	Units	Range of Detections for Detected Analytes that Exceed One or More Screening Levels	Range of Reporting Limits (RL) for Non-Detects	Ratio of Maximum Chemical Concentration to SL for Detected Analytes[1]	Screening Values Exceeded	Notes
DIBENZ(A,H)ANTHRACENE	RL	ug/kg	---	230	---	EPA RSL	---
INDENO(1,2,3-CD)PYRENE	RL	ug/kg	---	230	---	EPA RSL	---
N-NITROSODI-N-PROPYLAMINE	RL	ug/kg	---	230	---	EPA RSL	---
TRANS-1,4-DICHLORO-2-BUTENE	RL	ug/kg	---	7	---	EPA RSL	---

Notes:

[1] Ratio of maximum detected concentration to the SL is typically expressed as a Hazard Quotient (HQ). This ratio is presented above for each detected chemical and is calculated using the maximum detected concentration; the highest and lowest of screening values when multiple screening values are exceeded of same level in screening hierarchy. When more than two screening values are exceeded, the screening level used for calculation of the ratio (HQ) are in **bold**.

Screening Level Hierarchy for Human Health:

USEPA Residential RSLs (total carcinogenic and total non-carcinogenic), CHHSLs, and ODEQ bioaccumulation SLVs (Human Subsistence and Human General)

    a no ODEQ values

    b below USEPA RSLs, CHHSLs

    c ODEQ values not applicable per text of Appendix A (only applicable for J.C. Boyle Reservoir); USEPA RSL and CHHSLs not available

Key:		Units:	
EPA =	U.S. Environmental Protection Agency	metals:	mg/kg
RSL =	Residential Screening Level	pesticides:	ug/kg
TOT CAR =	Total carcinogen	dioxins and furans:	pg/g
TOT NON CAR =	Total non-carcinogen	SVOCs:	ug/kg
CHHSL =	California Human Health Screening Levels		
BSLV =	Land Quality Division Sediment Bioaccumulation Screening Level Values		
Comm =	commercial/industrial		
Res =	residential		

Table C-8. Chemicals in Sediment that Exceed One or More Marine Sediment Screening Levels. Source: CDM (2011).

Chemical	COC Based on Detect (D) or Elevated Reporting Limit (RL)	Units	Range of Detections for Detected Analytes that Exceed One or More Screening Levels	Range of Reporting Limits (RL) for Non-Detects	Ratio of Maximum Chemical Concentration to SL for Detected Analytes[a]	Screening Values Exceeded	Highest of Screening Value Hierarchy Level[b]
J.C. Boyle Reservoir							
Dieldrin	D	ug/kg	3.4	---	1.8	**SEF-SL1**, SEF-SL2, MS ERL, MS T20, MS TEL, MS T50, F-M	2a
2,3,4,7,8-PECDF	D	pg/g	1.5–1.5	---	1.4	ODEQ BSLV	2c
Butyl benzyl phthalate	RL	ug/kg	---	230–1,200	---	DMMP-ML	1
2,4-DIMETHYLPHENOL	RL	ug/kg	---	230–1,200	---	DMMP-ML	1
2-METHYLPHENOL	RL	ug/kg	---	230–1,200	---	DMMP-ML	1
4-METHYLPHENOL	RL	ug/kg	---	230–1,200	---	DMMP-ML	1
BENZOIC ACID	RL	ug/kg	---	930–4,800	---	DMMP-ML	1
BENZYL ALCOHOL	RL	ug/kg	---	230–1,200	---	DMMP-ML	1
HEXACHLOROBENZENE	RL	ug/kg	---	230–1,200	---	DMMP-ML	1
N-NITROSODIPHENYLAMINE	RL	ug/kg	---	230–1,200	---	DMMP-ML	1
Aroclor 1232	RL	ug/kg	---	0.16–0.24	---	DMMP-SL, SEF-SL1 (total PCBs)	2a
Aroclor 1242	RL	ug/kg	---	0.045–0.24	---	DMMP-SL, SEF-SL1 (total PCBs)	2a
Aroclor 1248	RL	ug/kg	---	0.045–0.24	---	DMMP-SL, SEF-SL1 (total PCBs)	2a
Aroclor 1254	RL	ug/kg	---	0.045–0.24	---	DMMP-SL, SEF-SL1 (total PCBs)	2a
Aroclor 1260	RL	ug/kg	---	0.045–0.24	---	DMMP-SL, SEF-SL1 (total PCBs)	2a
Chlordane (Technical)	RL	ug/kg	---	4.5–24	---	DMMP-SL, SEF-SL1	2a
Chlordane-Alpha	RL	ug/kg	---	0.9–4.9	---	DMMP-SL, SEF-SL1	2a
Chlordane-Gamma	RL	ug/kg	---	0.9–4.9	---	DMMP-SL, SEF-SL1	2a
Heptachlor	RL	ug/kg	---	0.9–4.9	---	SEF-SL1	2a
Endrin	RL	ug/kg	---	0.9–4.9	---	no value	2c
Heptachlor Epoxide	RL	ug/kg	---	0.9–4.9	---	SQuiRTs (T20, PEL)	2c
Toxaphene	RL	ug/kg	---	45–240	---	SQuiRTs (TEL)	2c
BENZO(B)FLUORANTHENE	RL	ug/kg	---	230–1200	---	SQuiRTs (T20, T50)	2c

Chemical	COC Based on Detect (D) or Elevated Reporting Limit (RL)	Units	Range of Detections for Detected Analytes that Exceed One or More Screening Levels	Range of Reporting Limits (RL) for Non-Detects	Ratio of Maximum Chemical Concentration to SL for Detected Analytes[a]	Screening Values Exceeded	Highest of Screening Value Hierarchy Level[b]
Butyl benzyl phthalate	RL	ug/kg		230–1,200		DMMP-ML	
Copco No. 1 Reservoir							
2,4-DIMETHYLPHENOL	RL	ug/kg	---	580–730	---	DMMP-ML	1
2-METHYLPHENOL	RL	ug/kg	---	580–730	---	DMMP-ML	1
BENZOIC ACID	RL	ug/kg	---	2,300–2,900	---	DMMP-ML	1
HEXACHLOROBENZENE	RL	ug/kg	---	580–730	---	DMMP-ML	1
N-NITROSODIPHENYLAMINE	RL	ug/kg	---	580–730	---	DMMP-ML	1
AROCLOR 1221	RL	ug/kg	---	0.24–0.3	---	DMMP-SL, SEF-SL1 (total PCBs)	2a
AROCLOR 1232	RL	ug/kg	---	0.12–0.15	---	DMMP-SL, SEF-SL1 (total PCBs)	2a
CHLORDANE (TECHNICAL)	RL	ug/kg	---	12–15	---	DMMP-SL, SEF-SL1	2a
CHLORDANE-ALPHA	RL	ug/kg	---	2.4–3	---	SEF-SL1	2a
CHLORDANE-GAMMA	RL	ug/kg	---	2.4–3	---	SEF-SL1	2a
DIELDRIN	RL	ug/kg	---	2.4–3	---	SEF-SL1	2a
HEPTACHLOR	RL	ug/kg	---	2.4–3	---	SEF-SL1	2a
HEPTACHLOR EPOXIDE	RL	ug/kg	---	2.4–3	---	SQuiRTs (T20, PEL)	2c
BENZO(B)FLUORANTHENE	RL	ug/kg	---	580–730	---	SQuiRTs (T20)	2c
BENZO(G,H,I)PERYLENE	RL	ug/kg	---	580–730	---	no value	2c
Iron Gate Reservoir							
1,4-DICHLOROBENZENE	RL	ug/kg	---	5–520	---	DMMP-ML	1
2,4-DIMETHYLPHENOL	RL	ug/kg	---	170–730	---	DMMP-ML	1
2-METHYLPHENOL	RL	ug/kg	---	170–730	---	DMMP-ML	1
BENZOIC ACID	RL	ug/kg	---	670–2900	---	DMMP-ML	1
HEXACHLOROBENZENE	RL	ug/kg	---	170–730	---	DMMP-ML	1
HEXACHLOROBUTADIENE	RL	ug/kg	---	5–520	---	DMMP-ML	1
N-NITROSODIPHENYLAMINE	RL	ug/kg	---	170–730	---	DMMP-ML	1
AROCLOR 1221	RL	ug/kg	---	0.067–0.3	---	DMMP-SL, SEF-SL1 (total PCBs)	2a

Chemical	COC Based on Detect (D) or Elevated Reporting Limit (RL)	Units	Range of Detections for Detected Analytes that Exceed One or More Screening Levels	Range of Reporting Limits (RL) for Non-Detects	Ratio of Maximum Chemical Concentration to SL for Detected Analytes[a]	Screening Values Exceeded	Highest of Screening Value Hierarchy Level[b]
AROCLOR 1232	RL	ug/kg	---	0.033–0.15	---	DMMP-SL, SEF-SL1 (total PCBs)	2a
CHLORDANE (TECHNICAL)	RL	ug/kg	---	3.3–15	---	DMMP-SL, SEF-SL1	2a
CHLORDANE-ALPHA	RL	ug/kg	---	0.67–3	---	SEF-SL1	2a
CHLORDANE-GAMMA	RL	ug/kg	---	0.67–3	---	SEF-SL1	2a
DIELDRIN	RL	ug/kg	---	0.67–3	---	SEF-SL1	2a
HEPTACHLOR	RL	ug/kg	---	0.67–3	---	SEF-SL1	2a
HEPTACHLOR EPOXIDE	RL	ug/kg	---	0.67–3	---	SEF-SL1	2a
1,2,4-TRICHLOROBENZENE	RL	ug/kg	---	5–520	---	DMMP-SL, SEF-SL1	2a
1,3-DICHLOROBENZENE	RL	ug/kg	---	5–520	---	DMMP-SL	2a
BENZO(B)FLUORANTHENE	RL	ug/kg	---	170–730	---	SQuiRTs (T20)	2c
Klamath River Estuary							
Lower Klamath							
2,4-DIMETHYLPHENOL	RL	ug/kg	---	230	---	DMMP-ML	1
2-METHYLPHENOL	RL	ug/kg	---	230	---	DMMP-ML	1
BENZOIC ACID	RL	ug/kg	---	910	---	DMMP-ML	1
N-NITROSODIPHENYLAMINE	RL	ug/kg	---	230	---	DMMP-ML	1
CHLORDANE (TECHNICAL)	RL	ug/kg	---	4.6	---	SEF-SL1	2a
HEPTACHLOR EPOXIDE	RL	ug/kg	---	0.91	---	SQuiRTs (T20)	2c
TOXAPHENE	RL	ug/kg	---	46	---	SQuiRTs (TEL)	2c
BENZO(B)FLUORANTHENE	RL	ug/kg	---	230	---	SQuiRTs (T20)	2c
Upper Klamath							
2,4-DIMETHYLPHENOL	RL	ug/kg	---	230	---	DMMP-ML	1
2-METHYLPHENOL	RL	ug/kg	---	230	---	DMMP-ML	1
BENZOIC ACID	RL	ug/kg	---	930	---	DMMP-ML	1
N-NITROSODIPHENYLAMINE	RL	ug/kg	---	230	---	DMMP-ML	1
CHLORDANE (TECHNICAL)	RL	ug/kg	---	4.6	---	SEF-SL1	2a

Chemical	COC Based on Detect (D) or Elevated Reporting Limit (RL)	Units	Range of Detections for Detected Analytes that Exceed One or More Screening Levels	Range of Reporting Limits (RL) for Non-Detects	Ratio of Maximum Chemical Concentration to SL for Detected Analytes[a]	Screening Values Exceeded	Highest of Screening Value Hierarchy Level[b]
HEPTACHLOR EPOXIDE	RL	ug/kg	---	0.93	---	SQuiRTs (T20)	2c
TOXAPHENE	RL	ug/kg	---	46	---	SQuiRTs (TEL)	2c
BENZO(B)FLUORANTHENE	RL	ug/kg	---	230	---	SQuiRTs (T20)	2c

Notes:
Screening Level Hierarchies for Marine Waters--
Retain if above:
1) DMMP-MLs
2a) SEF-SL1 or DMMP-SL
2b) SEF-SL1 or DMMP-SL AND SEF-SL2 or DMMP-BT
2c) Chemicals with no SEF or DMMP and one or more SQuiRTs exceeded
Key:

MS =	marine sediment	
ERL =	Effects Range Low	
TEL =	Threshold Effect Level	
SL1 =	Sediment Screening Level 1	
SEF =	Sediment Evaluation Framework	
T20 =	concentration representing 20 percent probability of observing effect	
T50 =	concentration representing 50 percent probability of observing effect	
BSLV =	Land Quality Division Sediment Bioaccumulation Screening Level Values	

Units:
metals: mg/kg
pesticides: ug/kg
dioxins and furans: pg/g
SVOCs: ug/kg
phthalates: ug/kg

DMMP = Dredged Material Management Program
ERM = Effects Range Median
F-M = Fish-marine
PEL = Probable Effect Level
ODEQ = Oregon Department of Environmental Quality
RL = Reporting Limit
D = Detect
ML = Maximum Level

[a] Ratio of maximum detected concentration to the SL is typically expressed as a Hazard Quotient (HQ). This ratio is presented above for each detected chemical and is calculated using the maximum detected concentration; the highest and lowest of screening values when multiple screening values are exceeded of same level in screening hierarchy. When more than two screening values are exceeded, the screening level used for calculation of the ratio (HQ) are in **bold**.
[b] Screening level hierarchy depicted on Figure 2 in CDM (2011).
Based on the information provided in Table A-5 in CDM (2011) and database query for ambiguous and positive exceedances.

Table C-9. 2009 and 2018 Pacific Northwest Sediment Evaluation Framework Freshwater Screening Levels. Adapted from RSET (2009) and RSET (2018).

Analyte	Units	Pacific Northwest SEF [a]				Add (+), Remove (-), or No Change (o) to COPC[d] list
		SL1[b] 2009	SL1[b] 2018	SL2[c] 2009	SL2[c] 2018	
Metals						
Arsenic	mg/kg	20	14	51	120	o
Cadmium	mg/kg	1.1	2.1	1.5	5.4	-
Chromium	mg/kg	95	72	100	88	o
Copper	mg/kg	80	400	830	1,200	o
Lead	mg/kg	340	360	430	>1,300	o
Mercury	mg/kg	0.28	0.66	0.75	0.8	o
Nickel	mg/kg	60	26	70	110	+
Selenium	mg/kg	n/a[e]	11	n/a	>20	o
Silver	mg/kg	2	0.57	2.5	1.7	o
Zinc	mg/kg	130	3,200	400	>4,200	o
Organics						
Polycyclic Aromatic Hydrocarbons (PAHs)						
Total PAHs	ug/kg	n/a	17,000	n/a	30,000	+
2-Methylnaphthalene	ug/kg	470	n/a	560	n/a	-
Acenaphthene	ug/kg	1,100	n/a	1,300	n/a	-
Acenaphthylene	ug/kg	470	n/a	640	n/a	-
Anthracene	ug/kg	1,200	n/a	1,600	n/a	-
Benz(a)anthracene	ug/kg	4,300	n/a	5,800	n/a	-
Benzo(a)pyrene	ug/kg	3,300	n/a	4,800	n/a	-
Benzo(g,h,i)perylene	ug/kg	4,000	n/a	5,200	n/a	-
Benzo(k)fluoranthene	ug/kg	600	n/a	4,000	n/a	-
Chrysene	ug/kg	5,900	n/a	6,400	n/a	-
Dibenz(a,h)anthracene	ug/kg	800	n/a	840	n/a	-
Dibenzofuran	ug/kg	400	n/a	440	n/a	-
Fluoranthene	ug/kg	11,000	n/a	15,000	n/a	-
Fluorene	ug/kg	1,000	n/a	3,000	n/a	-
Indeno(1,2,3-cd)pyrene	ug/kg	4,100	n/a	5,300	n/a	-
Naphthalene	ug/kg	500	n/a	1,300	n/a	-
Phenanthrene	ug/kg	6,100	n/a	7,600	n/a	-
Pyrene	ug/kg	8,800	n/a	16,000	n/a	-
Organics						
Polychlorinated biphenyl (PCBs)						
Total PCBs	pg/g	60,000	n/a	120,000	n/a	+
Total Aroclors	ug/kg	n/a	110	n/a	2,500	-

Analyte	Units	Pacific Northwest SEF [a]				Add (+), Remove (-), or No Change (o) to COPC[d] list
		SL1[b] 2009	SL1[b] 2018	SL2[c] 2009	SL2[c] 2018	
Organics						
Pesticides/Herbicides/Insecticides: Organochlorine Pesticides						
4,4'-DDD	ug/kg	n/a	310	n/a	860	o
4,4'-DDE	ug/kg	n/a	21	n/a	33	o
4,4'-DDT	ug/kg	n/a	100	n/a	8,100	o
BHC-alpha (HCH-alpha)	ug/kg	n/a	n/a	n/a	n/a	o
BHC-beta(HCH-beta)	ug/kg	n/a	n/a	n/a	n/a	o
BHC-gamma (HCH-gamma, Lindane)	ug/kg	n/a	n/a	n/a	n/a	o
Chlordane	ug/kg	n/a	n/a	n/a	n/a	o
Chlordane (technical)	ug/kg	n/a	n/a	n/a	n/a	o
Chlordane-alpha	ug/kg	n/a	n/a	n/a	n/a	o
Chlordane-gamma	ug/kg	n/a	n/a	n/a	n/a	o
Dieldrin	ug/kg	n/a	4.9	n/a	9.3	o
Heptachlor	ug/kg	n/a	n/a	n/a	n/a	o
Heptachlor epoxide	ug/kg	n/a	n/a	n/a	n/a	o
beta-Hexachlorocyclohexane	ug/kg	n/a	7.2	n/a	11	o
Endrin ketone	ug/kg	n/a	8.5	n/a	n/a	o
Organics **Phthalates**						
Bis(2-ethylhexyl) phthalate	ug/kg	220	500	320	22,000	o
Butyl benzyl phthalate	ug/kg	260	n/a	370	n/a	-
Dimethyl phthalate	ug/kg	46	n/a	440	n/a	-
Di-n-butyl-phthalate	ug/kg	n/a	380	n/a	1,000	+
Di-N-octyl phthalate	ug/kg	26	39	45	1,100	o
Organics **Semi-Volatile Organic Compounds (SVOCs): Phenols**						
Phenol	ug/kg	n/a	120	n/a	210	+
4-Methylphenol	ug/kg	n/a	260	n/a	2,000	+
Pentachlorophenol	ug/kg	n/a	1,200	n/a	<1,200	o
Organics **SVOCs: Chlorinated hydrocarbons**						
Hexachlorobenzene	ug/kg	n/a	n/a	n/a	n/a	o
Organics **Polychlorinated Dioxins and Furans**						
1,2,3,4,6,7,8-HPCDD	pg/g	n/a	n/a	n/a	n/a	o
1,2,3,4,6,7,8-HPCDF	pg/g	n/a	n/a	n/a	n/a	o
1,2,3,4,7,8,9-HPCDF	pg/g	n/a	n/a	n/a	n/a	o
1,2,3,4,7,8-HXCDD	pg/g	n/a	n/a	n/a	n/a	o
1,2,3,4,7,8-HXCDF	pg/g	n/a	n/a	n/a	n/a	o

Analyte	Units	Pacific Northwest SEF [a]				Add (+), Remove (-), or No Change (o) to COPC[d] list
		SL1[b] 2009	SL1[b] 2018	SL2[c] 2009	SL2[c] 2018	
1,2,3,6,7,8-HXCDD	pg/g	n/a	n/a	n/a	n/a	o
1,2,3,6,7,8-HXCDF	pg/g	n/a	n/a	n/a	n/a	o
1,2,3,7,8,9-HXCDD	pg/g	n/a	n/a	n/a	n/a	o
1,2,3,7,8,9-HXCDF	pg/g	n/a	n/a	n/a	n/a	o
1,2,3,7,8-PECDD	pg/g	n/a	n/a	n/a	n/a	o
1,2,3,7,8-PECDF	pg/g	n/a	n/a	n/a	n/a	o
2,3,4,6,7,8-HXCDF	pg/g	n/a	n/a	n/a	n/a	o
2,3,4,7,8-PECDF	pg/g	n/a	n/a	n/a	n/a	o
2,3,7,8-TCDD	pg/g	n/a	n/a	n/a	n/a	o
2,3,7,8-TCDF	pg/g	n/a	n/a	n/a	n/a	o
OCDD	pg/g	n/a	n/a	n/a	n/a	o
OCDF	pg/g	n/a	n/a	n/a	n/a	o
Site Specific Chemicals of Concern						
Butyltins						
Monobutyltin (µg/kg)	ug/kg	n/a	540	n/a	>4,800	o
Dibutyltin (µg/kg)	ug/kg	n/a	910	n/a	130,000	o
Tributyltin (µg/kg)	ug/kg	n/a	47	n/a	320	o
Tetrabutyltin (µg/kg)	ug/kg	n/a	97	n/a	>97	o
Total [Bulk] Petroleum Hydrocarbons (TPHs)						
TPH-diesel	mg/kg	n/a	340	n/a	510	+
TPH-residual	mg/kg	n/a	3,600	n/a	4,400	+
Miscellaneous Extractables Compounds						
Benzoic acid	ug/kg	n/a	2,900	n/a	3,800	+
Carbazole	ug/kg	n/a	900	n/a	1,100	+
Dibenzofuran	ug/kg	n/a	200	n/a	680	+

[a] Sediment Evaluation Framework
[b] Screening Level 1
[c] Screening Level 2
[d] Chemicals of Potential Concern
[e] not applicable
Units Key:
   g   gram
   kg   kilogram (1,000 grams)
   mg   milligram (10^{-3} grams)
   ug   microgram (10^{-6} grams)
   pg   picogram (10^{-12} grams)

Table C-10. Summary of the Lines of Evidence Used to Evaluate Each Exposure Pathway.
Source: CDM (2011).

Line of Evidence	Exposure Pathways				
	1	2	3	4	5
Sediment Evaluation Framework Level 2A Step 1 – Sediment Screening Levels					
1. DMMP Marine MLs				+	
Sediment Evaluation Framework Level 2A Steps 2a, 2b, 2c, 2d – Sediment Screening Levels					
2. Ecological SLs (freshwater and marine)			+	+	+
3. Ecological TEQ SLVs (sediment)			+	+	+
Sediment Evaluation Framework Level 2B – Results of Water Quality Criteria Evaluations and Bioassays					
4. Elutriate WQC (ecological)	+			+	
5. Benthic midge *(Chironomus dilutens)* Bioassay			+	+	+
6. Benthic amphipod *(Hyalella azteca)* Bioassay			+	+	+
7. Rainbow trout *(Oncorhynchus mykiss)* Bioassay	+			+	
8. Asian clam *(Corbicula fluminea)*[1] Bioaccumulation Study/BSAF			+		+
9. Blackworm *(Lumbriculus variegatus)*[2] Bioaccumulation Study/BSAF			+		+
10. Asian clam *(Corbicula fluminea)*[1] Tissue TRV			+	+	+
11. Blackworm *(Lumbriculus variegatus)*[2] Tissue TRV			+	+	+
Special Evaluations – Human Health in Sediment and Fish Tissue					
12. Yellow perch (*Perca flavescens*) Tissue TRV (ecological)			+	+	+
13. Bullhead (*Ameirus sp.*) Tissue TRV (ecological)			+	+	+
14. Fish Tissue TEQ (ecological)			+	+	+
15. HHSLs		+	+		+
16. HH TEQ SLVs (sediment)		+	+		+
17. Elutriate WQC (human health)					
18. Perch Tissue TRV (human health)			+		+
19. Bullhead Tissue TRV (human health)			+		+
20. Fish Tissue TEQ (human health)			+		+

+: Applicable line of evidence for exposure pathway

[1] Representative bivalve

[2] Representative oligochaete

Key:
  DMMP = Dredged Material Management Program
  ML =   Maximum Level
  SL =   Screening Level
  TEQ =   Toxic Equivalency
  SLV =   Screening Level Value
  WQC = Water Quality Criteria
  TRV =   Toxicity Reference Value
  BSAF = Biota-Sediment Accumulation Factor
  HHSL = Human Health Screening Level
  HH =   Human Health

Contaminants in Aquatic Biota

Separate assessments of contaminants in fish tissue for the Hydroelectric Reach have been undertaken by SWAMP and PacifiCorp. SWAMP data include sport fish tissue samples collected during 2007 and 2008 to evaluate accumulated contaminants in

nearly 300 lakes statewide. Sport fish were sampled to provide information on potential human exposure to selected contaminants and to represent the higher aquatic trophic levels (i.e., the top of the aquatic food web).

In the Hydroelectric Reach, fish tissue samples were collected in Copco No. 1 and Iron Gate reservoirs and analyzed for total mercury, selenium, and PCBs (Iron Gate Reservoir only) (Davis et al. 2010). SWAMP data for Iron Gate and Copco No. 1 reservoirs (Table C-11) indicate mercury tissue concentrations above the USEPA criterion of 300 nanograms per gram (ng/g) methylmercury in fish tissue to protect the health of consumers of noncommercial freshwater fish; and greater than the OEHHA public health guideline levels advisory tissue level (Klasing and Brodberg 2008) for consumption for 3 and 2 servings per week (70 and 150 ng/g wet weight, respectively) and the fish contaminant goal (220 ng/g wet weight). Measured selenium concentrations were 3 to 4 orders of magnitude lower than OEHHA thresholds of concern (2,500 to 15,000 ng/g wet weight) and PCB concentrations were below the lowest OEHHA threshold (i.e., fish contaminant goal of 3.6 ng/g wet weight) (Davis et al. 2010).

Table C-11. Total Mercury, Selenium, and PCBs in (ng/g wet weight) in Largemouth Bass taken from Iron Gate and Copco No. 1 Reservoirs During 2007 to 2008 (Davis et al. 2010).

Contaminant	Species	Iron Gate Reservoir	Copco No. 1 Reservoir
Methylmercury	Largemouth Bass (LMB)	330	310
Selenium	LMB	80	80
PCBs	LMB	1.31	*Not reported*

In a screening-level study of potential chemical contaminants in fish tissue in J.C. Boyle, Copco No. 1, and Iron Gate reservoirs, PacifiCorp analyzed metals (arsenic, cadmium, chromium, copper, lead, mercury, nickel, selenium, and zinc), organochlorine (pesticide) compounds, and PCBs in largemouth bass (*Micropterus salmoides*) (PacifiCorp 2004c). PacifiCorp reported that, in general, contaminant levels in fish tissue are below both screening level values for protection of human health (USEPA 2000) and recommended guidance values for the protection of wildlife (MacDonald 1994). Exceptions to this include measured fish tissue levels of total mercury in samples from Copco No. 1 and Iron Gate reservoirs as compared to the wildlife screening level of 0.00227 ug/g and measured fish tissue levels of arsenic (less than 0.3 ug/g) that PacifiCorp indicated may equal or exceed the toxicity screening level for subsistence fishers (0.147 ug/g) in samples of largemouth bass from J.C. Boyle, Copco No. 1, and Iron Gate reservoirs. Subsequent reanalysis of the PacifiCorp mercury tissue data indicates that all tissue samples exceed the most protective wildlife screening level of 0.00227 ug/g, samples from J.C. Boyle, Copco No. 1, and Iron Gate reservoirs exceed the screening level for subsistence fishers (0.049 ug/g), and samples from Copco No. 1 and Iron Gate reservoirs exceed the screening level for recreational fishers (0.4 ug/g) (Table C-12).

Table C-12. Total Mercury Concentrations (ug/g wet weight) in Largemouth Bass (LMB) Composite Tissue Samples taken from Lower Klamath Project Reservoirs in 2003 (PacifiCorp 2004c).

Sample	Composite	Site	Species	Total Mercury (ug/g wet weight)[1]
L-262-03	2F	J.C. Boyle Reservoir	LMB	0.153
L-262-03	3F	J.C. Boyle Reservoir	LMB	0.190
L-273-03	1F	Iron Gate Reservoir	LMB	0.564
L-273-03	2F	Iron Gate Reservoir	LMB	0.508
L-273-03	3F	Copco No. 1 Reservoir	LMB	0.563
L-273-03	4F	Copco No. 1 Reservoir	LMB	0.389
		Method Detection Limit		0.003[2]
		Method Reporting Limit		0.007[2]
		Screening Levels[3]:		
		Recreational fishers		0.4
		Subsistence fishers		0.049
		Wildlife		0.00227

[1] PacifiCorp (2004c) total mercury data was provided in ng/g dry weight. Data was converted to ug/g wet weight using percent moisture data provided for each sample by Moss Landing Marine Laboratory (A. Bonnema, pers. comm., 17 February 2011).

[2] The Method Detection Limit and Reporting Limit were converted from dry weight to wet weight using an average of the percent moisture data for all samples.

[3] Screening Levels (SLs) are numeric chemical guidelines that are used to assess and characterize the potential toxicity or bioaccumulative nature of environmental samples (i.e., sediments, water, organism tissue).

Additionally, PacifiCorp indicated that some of the fish tissue samples from J. C. Boyle and Copco No. 1 reservoirs exceeded the suggested wildlife screening value for total DDTs (Table C-13) (DDE,p,p' was detected; however DDT and DDD were not detected in the study), and total PCB values exceeded the screening level for subsistence fishers in largemouth bass from J.C. Boyle, Copco No. 1, and Iron Gate Reservoirs (Table C-14). Dioxins were not tested.

Table C-13. Total DDE Concentration (ng/g) in Large Mouth Bass (LMB) Composite Tissue Samples taken from Lower Klamath Project Reservoirs in 2003 (PacifiCorp 2004c).

Sample	Composite	Site	Species	DDE,p,p' (ng/g wet weight)
L-262-03	2F	J.C. Boyle Reservoir	LMB	<2.00
L-262-03	2F Duplicate	J.C. Boyle Reservoir	LMB	<2.00
L-262-03	3F	J.C. Boyle Reservoir	LMB	2.91
L-273-03	1F	Iron Gate Reservoir	LMB	<2.00
L-273-03	2F	Iron Gate Reservoir	LMB	<2.00
L-273-03	3F	Copco Reservoir	LMB	2.16
L-273-03	4F	Copco Reservoir	LMB	<2.00
		Method Detection Limit		0.56
		Method Reporting Limit		2
		Screening Levels[1] (for Total DDTs):		
		Recreational fishers		117
		Subsistence fishers		14.4
		Wildlife		0.2–1.07

[1] Screening Levels (SLs) are numeric chemical guidelines that are used to assess and characterize the potential toxicity or bioaccumulative nature of environmental samples (i.e., sediments, water, organism tissue).

Table C-14. Total PCB Concentrations (ng/g) in Large Mouth Bass (LMB) Composite Tissue Samples taken from Lower Klamath Project Reservoirs in 2003 (PacifiCorp 2004c).

Sample	Composite	Site	Species	Total PCB (ng/g wet weight)
L-262-03	2F	J.C. Boyle Reservoir	LMB	0.885
L-262-03	2F Duplicate	J.C. Boyle Reservoir	LMB	1.397
L-262-03	3F	J.C. Boyle Reservoir	LMB	3.521
L-273-03	1F	Iron Gate Reservoir	LMB	6.574
L-273-03	2F	Iron Gate Reservoir	LMB	4.909
L-273-03	3F	Copco Reservoir	LMB	2.822
L-273-03	4F	Copco Reservoir	LMB	2.158
		Method Detection Limit		Varies
		Method Reporting Limit		0.2
		Screening Levels[1]:		
		Recreational fishers		20
		Subsistence fishers		2.45
		Wildlife		100

[1] Screening Levels (SLs) are numeric chemical guidelines that are used to assess and characterize the potential toxicity or bioaccumulative nature of environmental samples (i.e., sediments, water, organism tissue).

To provide additional lines of evidence in the Klamath Dam Removal Secretarial Determination sediment evaluation, the potential for chemicals in sediment samples to bioaccumulate in aquatic species was investigated using laboratory invertebrates (Asian clams, *Corbicula fluminea*; and Black worms, *Lumbriculus variegates*) exposed to reservoir-derived sediments. Results indicated that multiple chemicals were found in

invertebrate tissue (acenaphthene, arsenic, benzo(a)pyrene, DDD/DDE, endosulfan I, endosulfan II, endosulfan sulfate, fluoranthene, hexachlorobenzene, lead, mercury, phenanthrene, pyrene, total PBDEs, total PCBs). Of these detected chemicals, only fluoranthene possessed a toxicity reference value (TRV) for the species tested; exceedances of the fluoranthene TRV were only identified above the No Effect TRV and were below the Low Effect TRV. Tissue-based TRVs were unavailable for the remaining invertebrate chemicals detected, and hexachlorobenzene has no tissue-based TRVs (for any species) (CDM 2011).

Lastly, two species of field-caught fish (yellow perch and bullhead) were collected during late September 2010 from J.C. Boyle, Copco No. 1, and Iron Gate reservoirs and analyzed for contaminant concentrations (CDM 2011). Table C-15 lists all the chemicals detected in the fish tissue along with a list of chemicals tested for, but not detected in any sample of fish tissue. Chemicals found in the fish tissue samples included 2,3,7,8-TCDD, arsenic, DDE/DDT, dieldrin, endrin, mercury, mirex, selenium, and total PCBs (CDM 2011). Mercury exceeded tissue-based TRVs for perch in Iron Gate Reservoir and bullhead samples in all three reservoirs (CDM 2011). TRVs were not available the remaining several chemicals detected in yellow perch and bullhead samples (CDM 2011). Results of chemical analyses of field collected fish revealed that no consistent pattern of contaminant distribution was identified among chemicals, media type, or location. Data revealed that fish can accumulate a fairly large number of sediment-associated chemicals; however, regional background conditions may be elevated for more than one of the measured chemicals (e.g., arsenic, mercury) (CDM 2011).

Table C-15. Chemicals Detected and Not Detected in Fish Tissue Field Caught in 2010 from J.C. Boyle, Copco No. 1, and Iron Gate reservoirs (CDM 2011).

Bullhead (*Ameiurus sp.*)		Yellow Perch (*Perca flavescens*)	
Detected[1]	Not Detected[2]	Detected[1]	Not Detected[2]
2,4'-DDD	2,4'-DDT	2,4'-DDD	Acenaphthene
2,4'-DDE	Acenaphthene	2,4'-DDE	Acenaphthylene
2-Fluorobiphenyl	Acenaphthylene	2,4'-DDT	Aldrin
4,4'-DDD	Aldrin	2-Fluorobiphenyl	Anthracene
4,4'-DDE	Anthracene	4,4'-DDD	BDE (4)
4,4'-DDT	BDE (3)	4,4'-DDE	Benzo(a)anthracene
alpha-BHC	Benzo(a)anthracene	4,4'-DDT	Benzo(a)pyrene
Arsenic	Benzo(a)pyrene	alpha-BHC	Benzo(b)fluoranthene
BDE (8)	Benzo(b)fluoranthene	Arsenic	Benzo(ghi)perylene
beta-BHC	Benzo(ghi)perylene	BDE (7)	Benzo(k)fluoranthene
cis-Chlordane	Benzo(k)fluoranthene	beta-BHC	Chrysene
cis-Nonachlor	Chrysene	cis-Chlordane	D/F (14)
D/F (1)	D/F (14)	cis-Nonachlor	Dibenz(a,h)anthracene
delta-BHC	Dibenz(a,h)anthracene	D/F (1)	Endosulfan I
Dieldrin	Endosulfan I	delta-BHC	Endosulfan II
Endosulfan sulfate	Endosulfan II	Dieldrin	Endrin aldehyde
Fluorene	Endrin	Endosulfan sulfate	Endrin ketone

Bullhead (*Ameiurus sp.*)		Yellow Perch (*Perca flavescens*)	
gamma-BHC (Lindane)	Endrin aldehyde	Endrin	Fluoranthene
Heptachlor epoxide	Endrin ketone	gamma-BHC (Lindane)	Fluorene
Hexachlorobenzene	Fluoranthene	Heptachlor	Indeno(1,2,3-cd)pyrene
Lead	Heptachlor	Heptachlor epoxide	Lead
Mercury	Indeno(1,2,3-cd)pyrene	Hexachlorobenzene	OCDF
Methoxychlor	OCDF	Mercury	PCB congeners (24)
Mirex	PCB congeners (28)	Methoxychlor	Phenanthrene
Naphthalene	Phenanthrene	Mirex	Pyrene
Nitrobenzene-d5	Pyrene	Naphthalene	Total HpCDF
OCDD	Total HpCDF	Nitrobenzene-d5	Total HxCDD
oxy-Chlordane	Total HxCDD	OCDD	Total HxCDF
PCB congeners (181)	Total HxCDF	oxy-Chlordane	Total PeCDD
Selenium	Total PeCDD	PCB congeners (185)	Total PeCDF
Total HpCDD	Total PeCDF	Selenium	Total TCDD
trans-Chlordane	Total TCDD	Total HpCDD	Total TCDF
trans-Nonachlor	Total TCDF	trans-Chlordane	
		trans-Nonachlor	

[1] Detected in one or more samples
[2] Not detected in any sample

In 2017, the State Water Board established water quality objectives for mercury to protect people and wildlife from consuming fish that contain high levels of mercury (State Water Board 2017). Five mercury fish tissue water quality objectives were developed depending on consumption patterns by individuals and wildlife. The Copco No. 1 and Iron Gate reservoirs were 303(d) listed as impaired for mercury by the State Water Board based on four or more lines of evidence showing mercury concentrations in fish fillets exceeding USEPA 304(a) concentrations of methylmercury in fish tissue of trophic level 4 fish (USEPA 2000, PacifiCorp 2004a, Davis et al. 2010, CDM 2011). No TMDL has been determined for Copco No. 1 or Iron Gate reservoirs, with an expected TMDL completion date of 2025.

C.7.2   Mid- to Lower Klamath Basin

C.7.2.1   Iron Gate Dam to Salmon River

Water Column Contaminants
SWAMP collected water quality data for inorganic and organic contaminants from 2000 through 2005 at eight monitoring sites from the Oregon-California state line (RM 214.1) to Klamath River at Klamath Glen (RM 5.9) (North Coast Regional Board 2008). As was the case for the SWAMP state line site (Section C.7.1.1), results for the four sites in the reach from Iron Gate Dam to the Salmon River indicated that with the exception of aluminum, all other measured concentrations of inorganic constituents (i.e., arsenic, cadmium, chromium, copper, lead, mercury, nickel, selenium, silver and zinc) were in compliance with all water quality objectives at the time of sampling. Aluminum

concentrations (26.3 to 280.0 ug/L) exceeded the USEPA continuous concentration for freshwater aquatic life protection (87 ug/L) on 23 of 59 site visits (39 percent exceedance rate), exceeded the USEPA secondary MCL for drinking water (50 ug/L) on 37 site visits (63 percent exceedance rate), and exceeded the California Department of Health Services secondary MCL for drinking water (200 ug/L) on five site visits (8 percent exceedances rate) (North Coast Regional Board 2008). The Klamath River from Iron Gate Dam to its confluence with the Scott River is 303(d) listed for aluminum. Water quality measurements in 2002 and 2003 at two USGS gage stations downstream of Iron Gate Dam to the Salmon River indicate that, with the exception of barium, nickel, magnesium, and calcium, the concentration of trace elements either remained the same or decreased as water flowed downstream, most likely because of binding to other particles and settling out of the water column (Tables C-16 and C-17; Flint et al. 2005). Additional data from two more USGS gage stations downstream of the confluence of the Klamath River and the Salmon River further support these trends, except for magnesium which does not continue to increase downstream of the Salmon River. Asarian and Kann (2014) note that the mean dissolved copper concentrations at three Klamath River sites between 2001 to 2013 are approximately 0.75 ug/L or less and the maximum is always less than 1.5 ug/L.

Sediment Contaminants

Sediment data for inorganic and organic contaminants in the Klamath River from Iron Gate Dam to the Salmon River are not readily available, nor are fish tissue analyses for contaminants in the lower Klamath River.

Table C-16. Water Quality Data Collected at 4 Sites in the Klamath River in 2002 (Flint et al. 2005).

Trace Element	Walker[1]	Seiad[2]	Orleans[3]	Klamath[4]
Aluminum	2	2	1	<1
Antimony	E0.04	E0.04	E0.05	<0.05
Arsenic	5	4	3	3
Barium	8.0	10.0	14.0	14.0
Beryllium	<0.06	<0.06	<0.06	<0.06
Cadmium	<0.04	<0.04	<0.04	<0.04
Chromium	<0.8	2.4	<0.8	<0.8
Cobalt	0.13	0.13	0.1	0.08
Copper	0.6	1.2	0.8	0.7
Manganese	6.1	7.2	2.5	3
Mercury	E0.01	0.01	<0.01	E0.01
Mercury (total recoverable)	E0.01	0.02	<0.01	<0.01
Molybdenum	1.0	1.0	0.8	0.6
Nickel	1.00	1.36	2.28	2.43
Selenium	<2	<2	<2	E1
Silver	<1	<1	<1	<1

Trace Element	Walker[1]	Seiad[2]	Orleans[3]	Klamath[4]
Zinc	ND	ND	ND	1
Uranium	0.13	0.12	0.11	0.07

Notes:
[1] Klamath River at Walker Bridge (USGS 11517818)
[2] Klamath River at Seiad Valley (USGS 11520500)
[3] Klamath River at Orleans (USGS 11523000)
[4] Klamath River near Klamath (USGS 11530500)
All data shown in micrograms per liter.

Table C-17. Water Quality Data Collected at 4 Sites in the Klamath River in 2003 (Flint et al. 2005).

Trace Element	Walker[1]		Seiad[2]		Orleans[3]		Klamath[4]	
	July	Sept	July	Sept	July	Sept	July	Sept
Calcium	15.3	12.7	16.9	14.4	16.0	15.6	15.5	15.9
Magnesium	9.63	8.21	10.6	9.22	8.19	8.69	7.59	8.77
Potassium	2.85	2.55	2.25	2.47	1.47	1.99	0.91	1.34
Sodium	16.6	14.1	13.4	13.9	8.18	11.3	5.25	7.59
Chloride	5.37	5.21	5.48	5.65	3.78	5.23	2.87	3.87
Fluoride	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2	<0.2
Silica	27.2	36.1	23.9	34.4	19.8	29.2	16.2	21.4
Sulfate	11.9	6.4	10.5	6.5	7.0	5.9	5.7	5.4

Notes:
[1] Klamath River at Walker Bridge (USGS 11517818)
[2] Klamath River at Seiad Valley (USGS 11520500)
[3] Klamath River at Orleans (USGS 11523000)
[4] Klamath River near Klamath (USGS 11530500[)]
All data shown in micrograms per liter.

C.7.2.2   Salmon River to Klamath River Estuary

Water Column Contaminants

SWAMP collected water quality data for inorganic and organic contaminants from 2001 through 2005 at three monitoring sites in this reach of the Klamath River to Klamath Glen (RM 5.9) (North Coast Regional Board 2008). With the exception of aluminum, all other measured concentrations of inorganic constituents (i.e., arsenic, cadmium, chromium, copper, lead, mercury, nickel, selenium, silver, and zinc) were in compliance with all water quality objectives at the time of sampling. Aluminum concentrations (8.8 to 565.0 ug/L) exceeded the USEPA continuous concentration for freshwater aquatic life protection (87 ug/L) on 12 of 28 site visits (43 percent exceedance rate), exceeded the USEPA secondary MCL for drinking water (50 ug/L) on 15 site visits (54 percent exceedance rate), and exceeded the California Department of Health Services secondary MCL for drinking water (200 ug/L) on four site visits (14 percent exceedances rate). At one station (Klamath River at Klamath Glen [RM 5.9]), grab samples were analyzed for 100 pesticides, pesticide constituents, or pesticide metabolites; 50 PCB congeners; and 6 phenolic compounds. There were no PCB detections, but the pesticide disulfoton was detected in one sample. Disulfoton is a systemic

organophosphate insecticide for which there is no numeric water quality objective. Similar to the Klamath River from downstream of Iron Gate to the confluence with the Salmon River, the water quality measurements in 2002 and 2003 at two USGS gage stations in the Lower Klamath River from the confluence with the Salmon River to the Klamath River Estuary indicate that, with the exception of barium, nickel, and calcium, the concentration of trace elements either remained the same or decreased as water flowed downstream, most likely because of binding to other particles and settling out of the water column (Tables C-16 and C-17; Flint et al. 2005). While barium concentrations remain the same at the two USGS gage stations between the Salmon River and the Klamath River Estuary (Orleans and Klamath), a comparison of barium concentrations at those gages with the two upstream gages (Walker Bridge and Seiad Valley) shows an increasing trend in barium with distance downstream from Iron Gate Dam. Nickel concentrations consistently increase with distance downstream.

Sediment Contaminants

Sediment data characterizing inorganic and organic contaminants in the Lower Klamath River from the Salmon River to the Klamath River Estuary are not readily available, nor are fish tissue analyses for contaminants in this reach of the Klamath River. Sediment data from the Klamath River Estuary are described below.

C.7.2.3   Klamath River Estuary

Sediment and water column data for inorganic and organic contaminants in the Klamath River Estuary are not readily available. However, contaminant conditions in the Klamath River Estuary (RM 0 to 3.9) are likely to be similar to those a few miles upstream at the site for which SWAMP data have been recently collected (see previous section).

As part of the Secretarial Determination studies, a sediment evaluation evaluated the potential environmental and human health impacts of the downstream release of sediment deposits currently stored behind the dams under the Proposed Project. Sediment cores were collected during 2009 to 2010 at multiple sites and at various sediment depths per site, including two locations in the Klamath River Estuary (see Section C.7.1.1). Overall, using thirteen lines of evidence from the 2009 to 2010 Secretarial Determination study (Lines of Evidence 1 to 11, 15, and 16 in Table C-10), sediment quality in the Klamath River Estuary does not appear to be highly contaminated (CDM 2011). The other lines of evidence were not used in the evaluation of conditions in the Klamath River Estuary, because no fish tissue was collected from the Klamath River Estuary. Where elevated concentrations of chemicals in sediment were found (i.e., arsenic, chromium, iron, nickel, bis[2-ethylhexyl]phthalate), the degree of exceedance based on comparisons of measured (i.e., detected) chemical concentrations to SLs was small and in several cases (i.e., arsenic, nickel) may reflect regional background conditions (CDM 2011). The results of the acute toxicity bioassays for midge and amphipod identified no statistically significant difference in survival of either test organism exposed to estuary sediments compared to control sediments. As with the reservoir sediments (Section C.7.1.1), the lone chemical identified in tissue from invertebrates exposed to estuary sediments above TRVs was fluoranthene. Further, it was only identified above the No Effect TRV, and was below the Low Effect TRV. TEQs for dioxin, furan, and dioxin-like PCBs were all below 0.2 pg/g for the Klamath River Estuary, thus adverse effects from exposure to TEQs are not expected following exposure to sediment in the Klamath River Estuary (CDM 2011).

C.8   References

Armour, C. L. 1991. Guidance for evaluating and recommending temperature regimes to protect fish. USFWS Biological Report 90 (22).

Armstong, N., and G. Ward. 2008. Coherence of nutrient loads and AFWO Klamath River grab sample water quality database. Technical Report. Prepared for USFWS, Arcata Fish and Wildlife Office, Arcata, California.

Asarian, E., and J. Kann. 2006a. Evaluation of PacifiCorp's Klamath River water quality model predictions for selected water quality parameters. Technical Memorandum. Prepared by Kier Associates, Blue Lake and Arcata, California and Aquatic Ecosystem Sciences, LLC, Ashland, Oregon for the Yurok Tribe Environmental Program, Klamath, California.

Asarian, E., and J. Kann. 2006b. Klamath River nitrogen loading and retention dynamics, 1996–2004. Final Technical Report Prepared by Kier Associates, Blue Lake and Arcata California and Aquatic Ecosystem Sciences, LLC, Ashland, Oregon for the Yurok Tribe Environmental Program, Klamath, California.

Asarian, E., and J. Kann. 2011. Phytoplankton and Nutrient Dynamics in Iron Gate and Copco No. 1 Reservoirs 2005–2010. Prepared by Kier Associates, Eureka, California and Aquatic Ecosystem Sciences, LLC, Ashland, Oregon for the Yurok Tribe Environmental Program, Klamath, California.

Asarian, E., and J. Kann. 2013. Synthesis of Continuous Water Quality Data for the Lower and Middle Klamath River, 2001–2011. Prepared by Kier Associates, Eureka, California and Aquatic Ecosystem Sciences, LLC, Ashland, Oregon for the Klamath Basin Tribal Water Quality Work Group.

Asarian, E., J. Kann, and W. W. Walker. 2009. Multi-year nutrient budget dynamics for Iron Gate and Copco Reservoirs, California. Prepared by Riverbend Sciences and Kier Associates, Eureka, California, Aquatic Ecosystem Sciences, LLC, Ashland, Oregon, and William Walker, Concord, Massachusetts for the Karuk Tribe, Department of Natural Resources, Orleans, California.

Asarian, E., J. Kann, and W.W. Walker. 2010. Klamath River nutrient loading and retention dynamics in free-flowing reaches, 2005–2008. Prepared by Kier Associates, Eureka, California and Aquatic Ecosystem Sciences, LLC, Ashland, Oregon for the Yurok Tribe Environmental Program, Klamath, California.

Asarian, J. E., Y. Pan, N.D. Gillett, and J. Kann. 2015. Periphyton assemblages and associated environmental conditions in the Klamath River 2004 -2013. Prepared by Riverbend Sciences, Portland State University, and Aquatic Ecosystem Sciences, LLC. for the Klamath Basin Tribal Quality Group.

Barnes, R. 1980. Coastal lagoons, the natural history of a neglected habitat. Cambridge University Press, United Kingdom.

Bartholow, J. 2005. Recent water temperature trends in the lower Klamath River, California. North American Journal of Fisheries Management 25: 152–162.

Basdekas, L., and M. Deas. 2007. Technical Memorandum No. 7 Temperature and flow dynamics of the Klamath River. Submitted to the Bureau of Reclamation Klamath Basin Area Office. Cramer Fish Sciences.
http://www.fishsciences.net/projects/klamathcoho/tech_memo_7.php

Blakey, J.F. 1966. Temperature of surface waters in the conterminous United States. Hydrologic Investigations Atlas HA-235. U.S. Geological Survey, Washington, D.C.
http://ut.water.usgs.gov/publications/pubsmaps.html.

Brett, J.R. 1952. Temperature tolerance in young Pacific salmon, genus *Oncorhynchus.* Journal of the Fisheries Research Board of Canada 9: 265–323.

Bruun, P. 1966. Tidal inlets and littoral drift. Universiteforlaget, Oslo, Norway.

Butcher, J. 2008. Nutrient dynamics in the Klamath. Memorandum from J. Butcher, Tetra Tech, Inc., Research Triangle Park, North Carolina to the Klamath TMDL Technical Team. 12 February.

Carlson, K., and K. Foster. 2008. Water quality monitoring during turbine venting tests at the Iron Gate Powerhouse, Klamath Hydroelectric Project. Final Report. Prepared by CH2M HILL and Mason, Bruce, and Girard for PacifiCorp Energy, Portland, Oregon.

CDM. 2011. Screening-level evaluation of contaminants in sediments from three reservoirs and the estuary of the Klamath River, 2009–2011. Prepared for the U.S. Department of the Interior, Klamath River Secretarial Determination, Water Quality Sub Team, Sacramento, California. Available at: http://klamathrestoration.gov/.

CH2M Hill. 2008. Blue-green algae (cyanobacteria) and microcystin monitoring results in the vicinity of the Klamath Hydroelectric Project: September 9-11, 2008. Technical Memorandum. Prepared by Ken Carlson (CH2M HILL) and Richard Raymond (E&S Environmental Chemistry) for Cory Scott and Linda Prendergast, PacifiCorp, Portland, Oregon.

CH2M Hill. 2009. Analysis of microcystin in resident fish and mussel tissues in the vicinity of the Klamath Hydroelectric Project in 2008. Prepared by CH2M HILL Inc. Portland, Oregon for PacifiCorp Energy, Portland, Oregon.

Davis, J. A., A. R. Melwani, S. N. Bezalel, J. A. Hunt, G. Ichikawa, A. Bonnema, W. A. Heim, D. Crane, S. Swensen, C. Lamerdin, and M. Stephenson. 2010. Contaminants in fish from California lakes and reservoirs, 2007–2008. Summary report on a two-year screening survey. Prepared for Surface Water Ambient Monitoring Program, California State Water Resources Control Board, Sacramento, California.

Deas, M. 2008. Nutrient and organic matter fate and transport in the Klamath River: June to September 2007. Prepared by Watercourse Engineering, Inc., Davis, California for PacifiCorp, Portland, Oregon.

Deas, M. L., and G. T. Orlob. 1999. Klamath River modeling project. Report No. 99-04. Prepared by Center for Environmental and Water Resources Engineering, Department

of Civil and Environmental Engineering, Water Resources Modeling Group, University of California, Davis.

Deas, M., and J. Vaughn. 2006. Characterization of organic matter fate and transport in the Klamath River below Link Dam to assess treatment/reduction potential. Prepared by Watercourse Engineering, Inc., Davis, California for the U.S. Bureau of Reclamation, Klamath Area Office, California.

Dillon, J. 2008. Subject: Dioxin in sediments behind the dams on the Klamath River. Technical memorandum to B. Cluer, Team Leader, Scientific Support Team and S. Edmondson, Northern California Habitat Supervisor, from J. Dillon, Water Quality Program Coordinator, National Marine Fisheries Service, Southwest Region, Santa Rosa, California. 8 April 2008.

Doyle, M. C., and D. D. Lynch. 2005. Sediment oxygen demand in Lake Ewauna and the Klamath River, Oregon, June 2003. Scientific Investigations Report 2005-5228. Prepared by the U.S. Department of the Interior, U.S. Geological Survey, Reston, Virginia in cooperation with the Bureau of Reclamation.

Dunsmoor, L. K., and C. W. Huntington. 2006. Suitability of environmental conditions within Upper Klamath Lake and the migratory corridor downstream for use by anadromous salmonids. Technical Memorandum to the Klamath Tribes. Prepared for the Klamath Tribes, Chiloquin, Oregon.

Eilers, J. M., and C. P. Gubala. 2003. Bathymetry and sediment classification of the Klamath Hydropower Project impoundments. Prepared by J. C. Headwaters, Inc. for PacifiCorp, Portland, Oregon.

Eilers, J. M., and R. Raymond. 2005. Appendix E. Sediment oxygen demand in selected sites of the Lost River and Klamath River. Prepared by MaxDepth Aquatics, Inc., Bend, Oregon, and Environmental Science Resources, LLC, Corvallis, Oregon for Tetra Tech, Inc., Fairfax, Virginia.

E&S Environmental Chemistry, Inc. 2017. Results of cyanobacteria and microcystin monitoring in the vicinity of the Klamath Hydroelectric Project. Technical Memorandum. Prepared by E&S Environmental Chemistry, Inc. for PacifiCorp, Portland, Oregon.

Escoffier, F. F. 1940. The stability of tidal inlets. Shore and Beach 8: 114–115.

Fetcho, K. 2006. Klamath River blue-green algae summary report. Water Year 2005. Prepared by the Yurok Tribe Environmental Program, Klamath, California.

Fetcho, K. 2007. 2006 Klamath River blue-green algae summary report. Prepared by the Yurok Tribe Environmental Program, Klamath, California.

Fetcho, K. 2008. FINAL 2007 Klamath River blue-green algae summary report. Prepared by the Yurok Tribe Environmental Program, Klamath, California.

Fetcho, K. 2009. FINAL 2008 Klamath River blue-green algae summary report. Prepared by the Yurok Tribe Environmental Program, Klamath, California.

Fetcho, K. 2010. September 14 and 15 phytoplankton results and recent microcystin results. Memorandum from the Yurok Tribe Environmental Program, Klamath, California.

Fetcho, K. 2011. FINAL 2009 Klamath River blue-green algae summary report. Prepared by the Yurok Tribe Environmental Program, Klamath, California.

FERC (Federal Energy Regulatory Commission). 2007. Final Environmental Impact Statement for hydropower license. Klamath Hydroelectric Project (FERC Project No. 2082-027). http://www.ferc.gov/industries/hydropower/enviro/eis/2007/11-16-07.asp.

FISHPRO. 2000. Fish passage conditions on the upper Klamath River. Submitted to Karuk Tribe and PacifiCorp.

Flint, L. E., A. L. Flint, D. S. Curry, S. A. Rounds, and M. C. Doyle. 2005. Water-Quality Data from 2002 to 2003 and Analysis of Data Gaps for Development of Total Maximum Daily Loads in the Lower Klamath River Basin, California. U. S. Geological Survey Scientific Investigations Report 2004-5255.

GEC (Gathard Engineering Consulting). 2006. Klamath River dam and sediment investigation. Prepared by GEC, Seattle, Washington.

Gibson, M. 2016. 2010-2015 Klamath River Blue-Green Algae Summary Report. Final Report. Prepared by Yurok Tribe Environmental Program: Water Division, Klamath, California.

Gray, J. R., G. D. Glysson, L. M. Turcios, and G. E. Schwarz. 2000. Comparability of suspended-sediment concentration and total suspended solids data. Water Resources Investigations Report 00-4191. U.S. Geological Survey, Reston, Virginia. Available at: http://water.usgs.gov/pubs/wri/wri004191.

Hanington, M. 2013. 2012 Klamath River continuous water quality monitoring summary report. Final Report. Prepared by Yurok Tribe Environmental Program: Water Division, Klamath, California.

Hanington, M. and K. Torso. 2013. 2012 Klamath River nutrient summary report. Final Report. Prepared by Yurok Tribe Environmental Program: Water Division, Klamath, California.

Hanington, M., and K. Ellien. 2013. 2013 Klamath River continuous water quality monitoring summary report. Final Report. Prepared by Yurok Tribe Environmental Program: Water Division, Klamath, California.

Hanington, M. and S. Stawasz. 2014. 2013 Klamath River nutrient summary report. Final Report. Prepared by Yurok Tribe Environmental Program: Water Division, Klamath, California.

Hanington, M. and R. Cooper-Carouseli. 2014. 2014 Klamath River nutrient summary report. Final Report. Prepared by Yurok Tribe Environmental Program: Water Division, Klamath, California.

Hiner, M. 2006. Seasonal water quality in the Klamath River estuary and surrounding sloughs, 2001–2003.

Horne, A. and Goldman, C. 1994. Limnology. McGraw-Hill, New York.

HVTEPA (Hoopa Valley Tribe Environmental Protection Agency). 2008. Water quality control plan Hoopa Valley Indian Reservation. Approved 11 September 2002, Amendments Approved 14 February 2008. Hoopa Valley Tribal Environmental Protection Agency, Hoopa, California.

HVTEPA. 2013. Water Quality Monitoring by the Hoopa Tribal Environmental Protection Agency 2008-2012. Prepared by the Hoopa Tribal Environmental Protection Agency in cooperation with Kier Associates. Hoopa Valley Tribal Environmental Protection Agency, Hoopa, California.

Jacoby, J.M. and J. Kann. 2007. The occurrence and response to toxic cyanobacteria in the Pacific Northwest, North America. Lake and Reservoir Management 23: 123–143.

Kann, J. 2004. Memo: Copco Lake analysis. Letter to Kier and Associates from J. Kann, Aquatic Ecologist, Aquatic Ecosystem Sciences, LLC, Ashland, Oregon. 7 December.

Kann, J. 2006. *Microcystis aeruginosa* occurrence in the Klamath River system of southern Oregon and northern California. Technical Memorandum.

Kann, J. 2007a. Toxic cyanobacteria results for Copco/Iron Gate reservoirs: 31 May and 12–13 June 2007. Technical memorandum from J. Kann, Aquatic Ecologist, Aquatic Ecosystem Sciences, LLC, Ashland, Oregon. 19 June.

Kann, J. 2007b. Copco/Iron Gate reservoir toxic cyanobacteria results: June 26-27th, 2007. Technical memorandum from J. Kann, Aquatic Ecologist, Aquatic Ecosystem Sciences, LLC, Ashland, Oregon. 29 June.

Kann, J. 2007c. Toxic cyanobacteria results for Copco/Iron Gate reservoirs: 10–11 July 2007. Technical memorandum from J. Kann, Aquatic Ecologist, Aquatic Ecosystem Sciences, LLC, Ashland, Oregon. 16 July.

Kann, J. 2007d. Toxic cyanobacteria results for Copco/Iron Gate reservoirs: August 7-8, 2007. Technical memorandum from J. Kann, Aquatic Ecologist, Aquatic Ecosystem Sciences, LLC, Ashland, Oregon. 15 August.

Kann, J. 2008a. Microcystin bioaccumulation in Klamath River fish and freshwater mussel tissue: preliminary 2007 results. Technical Memorandum. Prepared by Aquatic Ecosystem Sciences, LLC, Ashland, Oregon for the Karuk Tribe of California, Orleans, California.

Kann, J. 2008b. Expert report of Jacob Kann, Ph.D. in the matter of McConnell et al. v. PacifiCorp, Inc. Prepared by Aquatic Ecosystem Sciences, LLC, Ashland, Oregon for Lawyers for Clean Water.

Kann, J. 2014. Evaluation of cyanobacteria and cyanobacterial toxins with reference to selection of water quality criteria for the Karuk Tribe of California. Technical Memorandum. Prepared by Aquatic Ecosystem Sciences, LLC, Ashland, Oregon for the Klamath Tribe of California, Natural Resources Department, Orleans, California.

Kann, J. and E. Asarian. 2005. 2002 Nutrient and hydrologic loading to Iron Gate and Copco reservoirs, California. Prepared by Aquatic Ecosystem Sciences, LLC, Ashland, Oregon and Kier Associates, Mill Valley and Arcata, California for the Karuk Tribe of California, Department of Natural Resources, Orleans, California.

Kann, J. and E. Asarian. 2006. Longitudinal analysis of Klamath River phytoplankton data 2001–2004. Technical Memorandum. Prepared by Aquatic Ecosystem Sciences, LLC, Ashland, Oregon and Kier Associates, Blue Lake and Arcata, California for the Yurok Tribe Environmental Program, Klamath, California.

Kann, J. and E. Asarian. 2007. Nutrient budgets and phytoplankton trends in Iron Gate and Copco reservoirs, California, May 2005–May 2006. Prepared by Aquatic Ecosystem Sciences, LLC, Ashland, Oregon and Kier Associates, Arcata, California and by the Karuk Tribe of California, Department of Natural Resources, Orleans, California for the State Water Resources Control Board.

Kann, J. and C. Bowman. 2012. Middle Klamath River toxic cyanobacteria trends, 2010. Technical Memorandum. Prepared by Aquatic Ecosystem Sciences, LLC, Ashland, Oregon and Karuk Tribe Department of Natural Resources, Orleans, California for the Karuk Tribe Department of Natural Resources, Orleans, California.

Kann, J. and S. Corum. 2006. Summary of 2005 toxic *Microcystis aeruginosa* trends in Copco and Iron Gate reservoirs on the Klamath River, California. Technical Memorandum. Prepared by Aquatic Ecosystem Sciences, LLC, Ashland, Oregon and the Karuk Tribe Department of Natural Resources for the Karuk Tribe Department of Natural Resources, Orleans, California.

Kann, J. and S. Corum. 2007. Summary of 2006 toxic *Microcystis aeruginosa* and microcystin trends in Copco and Iron Gate reservoirs, California. Technical Memorandum. Prepared by Aquatic Ecosystem Sciences, LLC, Ashland, Oregon and the Karuk Tribe Department of Natural Resources for the Karuk Tribe Department of Natural Resources, Orleans, California.

Kann, J. and S. Corum. 2009. Toxigenic *Microcystis aeruginosa* bloom dynamics and cell density/chlorophyll a relationships with microcystin toxin in the Klamath River, 2005–2008. Technical Memorandum. Prepared by Aquatic Ecosystem Sciences, LLC, Ashland, Oregon and the Karuk Tribe Department of Natural Resources for the Karuk Tribe Department of Natural Resources, Orleans, California.

Kann, J. and W. W. Walker. 1999. Nutrient and hydrologic loading to Upper Klamath Lake, Oregon, 1991–1998. Draft Report. Prepared by Aquatic Ecosystem Sciences, LLC, Ashland, Oregon for the Klamath Tribes Natural Resources Department, U.S. Bureau of Reclamation Cooperative Studies.

Kann, J., L. Bowater, and S. Corum. 2010a. Middle Klamath River toxic cyanobacteria trends, 2009. Technical Memorandum. Prepared by Aquatic Ecosystem Sciences, LLC,

Ashland, Oregon and the Karuk Tribe Department of Natural Resources for the Karuk Tribe Department of Natural Resources, Orleans, California.

Kann, J., S. Corum, and K. Fetcho. 2010b. Microcystin bioaccumulation in Klamath River freshwater mussel tissue: 2009 results. Prepared by Aquatic Ecosystem Sciences, LLC, for the Karuk Tribe Natural Resources Department, and the Yurok Tribe Environmental Program.

Kann, J., L. Bowater, G. Johnson, and C. Bowman. 2011. Preliminary 2010 microcystin bioaccumulation results for Klamath River salmonids. Technical Memorandum. Prepared by Aquatic Ecosystem Sciences LLC for the Karuk Tribe Department of Natural Resources, Orleans California.

Kann, J., C. Bowman, L. Bowater, G. Johnson, and S. Raverty. 2013. Microcystin bioaccumulation in Klamath River salmonids; 2010 Study Results (Updated 6-12-2013). Technical memorandum. Prepared by Aquatic Ecosystem Sciences for the Karuk Tribe Department of Natural Resources, Orleans California.

Karuk Tribe of California. 2001. Karuk aboriginal territories Indian Creek and Elk Creek water quality monitoring report for the fall 2000 monitoring period. Prepared by the Karuk Tribe of California, Department of Natural Resources, Orleans, California.

Karuk Tribe of California. 2002. Water quality monitoring report, Water Year 2000 and 2001. Karuk Tribe of California, Water Resources, Department of Natural Resources, Orleans, California.

Karuk Tribe of California. 2003. Water quality monitoring report, Water Year 2002. Karuk Tribe of California, Water Resources, Department of Natural Resources, Orleans, California.

Karuk Tribe of California. 2007. 2007 Water quality assessment report for Klamath River, Salmon River, Scott River, Shasta River, Ti-Bar Creek, and Irving Creek. Prepared by Karuk Tribe of California, Water Resources, Department of Natural Resources, Orleans, California.

Karuk Tribe of California. 2009. 2008 Water quality assessment report for Klamath River, Salmon River, Scott River, Shasta River, and Bluff Creek. Prepared by Karuk Tribe of California, Water Quality, Department of Natural Resources, Orleans, California.

Karuk Tribe of California. 2010a. Water quality report for the mid-Klamath, Salmon, Scott, and Shasta rivers: May–December 2009. Prepared by Karuk Tribe of California, Water Quality Program, Department of Natural Resources, Orleans, California.

Karuk Tribe of California. 2010b. 2010 Water quality assessment report: Klamath River, Salmon River, Scott River, Shasta River, and Bluff Creek. Prepared by Karuk Tribe of California, Water Quality Program, Department of Natural Resources, Orleans, California.

Karuk Tribe of California. 2011. 2011 Water quality assessment report: Klamath River, Salmon River, Scott River, Shasta River, and Bluff Creek. Prepared by Karuk Tribe of

California, Water Quality Program, Department of Natural Resources, Orleans, California.

Karuk Tribe of California. 2012. 2012 Water quality assessment report: Klamath River, Salmon River, Scott River, Shasta River, and Camp Creek. Prepared by Karuk Tribe of California, Water Quality Program, Department of Natural Resources, Orleans, California.

Karuk Tribe of California. 2013. 2013 Water quality assessment report: Klamath River, Salmon River, Scott River, Shasta River, and Camp Creek. Prepared by Karuk Tribe of California, Water Quality Program, Department of Natural Resources, Orleans, California. Available at: http://karuk.us/index.php/departments/natural-resources/somes-bar-water-quality.

Kier Associates. 2006. Appendix G. Nutrient criteria for the Klamath River on the Hoopa Valley Indian Reservation. Hoopa Valley Indian Reservation water quality control plan. Prepared by Kier Associates, Mill Valley and Arcata, California for the Hoopa Valley Tribal Environmental Protection Agency.

Kirk, S., D. Turner, and J. Crown. 2010. Upper Klamath and Lost River sub-basins total maximum daily load (TMDL) and water quality management plan (WQMP). Oregon Department of Environmental Quality, Bend, Oregon.

Klasing, S., and R. Brodberg. 2008. Development of fish contaminant goals and advisory tissue levels for common contaminants in California sport fish: chlordane, DDTs, dieldrin, methylmercury, PCBs selenium, and toxaphene. Prepared by Pesticide and Environmental Toxicology Branch, Office of Environmental Health Hazard Assessment, California Environmental Protection Agency.

KRRC (Klamath River Renewal Corporation). 2018. Draft water temperature data collection at Shovel Creek. Prepared by KRRC, San Francisco, California for State Water Resources Control Board, Sacramento, California.

KTWQC (Klamath Tribal Water Quality Consortium). 2016. Upper Klamath Basin nonpoint source pollution assessment and management program plan. Prepared with assistance from Kier Associates and Riverbend Sciences for the Klamath Tribal Water Quality Consortium.

Lytle, C. M. 2000. Subject: Water quality data review and wetland size estimate for the treatment of wastewaters from the Klamath Straits Drain. Draft Technical Memorandum. Prepared for USBLM, Klamath Project Office, Klamath Falls.

MacDonald, L.H. 1994. Developing a monitoring project. Journal of Soil and Water Conservation May–June: 221–227.

MacDonald, L.H, A. W. Smart, and R. C. Wissmar. 1991. Monitoring guidelines to evaluate effects of forestry activities on streams in the Pacific Northwest and Alaska. Report No. 910/9-91-001. U.S. Environmental Protection Agency, Region 10, Seattle, Washington and Center for Streamside Studies, University of Washington, Seattle.

Mayer, T.D. 2005. Water quality impacts of wetland management in the Lower Klamath National Wildlife Refuge, Oregon and California, USA. Wetlands 25: 697–712.

McGeer, J.C., L. Baranyi, and G. K. Iwama. 1991. Physiological responses to challenge tests in six stocks of coho salmon (*Oncorhynchus kisutch*). Canadian Journal of Fisheries and Aquatic Sciences 48: 1,761–1,771.

Miller, M.A., R. M. Kudela, A. Mekebri, D. Crane, S. C. Oates, M. T. Tinker, M. Staedler, W. A. Miller, S. Toy-Choutka, C. Dominik, D. Hardin, G. Langlois, M. Murray, K. Ward, D. A. Jessup. 2010. Evidence for a novel marine harmful algal bloom: cyanotoxin (microcystin) transfer from land to sea otters. PLoS ONE 5: e12576. doi:10.1371/journal.pone.0012576.

Montgomery, J.M. 1985. Water treatment principles and design. J. M. Montgomery Consulting Engineers. John Wiley and Sons, Inc.

NMFS (National Marine Fisheries Service). 2002. Biological opinion Klamath Project Operations.

NMFS and USFWS (National Marine Fisheries Service and U.S. Fish and Wildlife Service). 2013. Biological opinions on the effects of Proposed Klamath Project Operations from May 31, 2013, through March 31, 2023, on five federally listed threatened and endangered species NMFS File Number SWR-2012-9371 and FWS File Number 08EKLA00-2013-F-0014.

North Coast Regional Board (North Coast Regional Water Quality Control Board). 2008. Surface water ambient monitoring program (SWAMP) summary report for the North Coast Region (RWQCB-1) for years 2000–2006. Prepared by North Coast Regional Water Quality Control Board, Santa Rosa, California.

North Coast Regional Board. 2010. Klamath River total maximum daily loads (TMDLs) addressing temperature, dissolved oxygen, nutrient, and microcystin impairments in California, the proposed site specific dissolved oxygen objectives for the Klamath River in California, and the Klamath River and Lost River implementation plans. Final Staff Report. North Coast Regional Water Quality Control Board, Santa Rosa, California.

North Coast Regional Board. 2011. Water Quality Control Plan for the North Coast region (Basin Plan). Santa Rosa, California.

North Coast Regional Board. 2013. State and U.S. EPA warns to avoid contact with blue-green algae in Iron Gate and Copco Reservoirs. Media Release. California Regional Water Quality Control Board, North Coast Region, Santa Rosa, California.

North Coast Regional Board. 2014. Blue-green algae blooms in Klamath River and Reservoirs result in warning against water contact or use. Media Release. California Regional Water Quality Control Board, North Coast Region, Santa Rosa, California.

North Coast Regional Board. 2016. Blue-green algae blooms result in warning against water contact or use at Iron Gate and Copco Reservoirs on the Klamath River. Media Release. California Regional Water Quality Control Board, North Coast Region, Santa Rosa, California.

North Coast Regional Board. 2017. Blue-green algae bloom triggers warning against water contact at Copco Reservoir on the Klamath. Media Release. California Regional Water Quality Control Board, North Coast Region, Santa Rosa, California.

NRC (National Research Council). 2003. Endangered and threatened fishes in the Klamath River basin, causes of decline and strategies for recovery. Prepared by the Committee on Endangered and Threatened Fishes in the Klamath River Basin, Board on Environmental Studies and Toxicology, Division on Earth and Life Studies, National Research Council of the National Academies. The National Academies Press, Washington, D.C.

O'Brien, M.P. 1971. Notes on tidal inlets on sandy shores. Hydraulic Engineering Lab Report No. HEL-24-5. University of California.

ODEQ (Oregon Department of Environmental Quality) 2002. Upper Klamath Lake drainage total maximum daily load (TMDL) and water quality management plan (WQMP). Portland, Oregon.

ODEQ. 2013. Development of Oregon background metals concentrations in soil. Technical Report. Prepared by Oregon Department of Environmental Quality, Portland, Oregon. https://www.oregon.gov/deq/FilterDocs/DebORbackgroundMetal.pdf.

Oliver A. A., R. A. Dahlgren, and M. L. Deas. 2014. The upside-down river: Reservoirs, algal blooms, and tributaries affect temporal and spatial patterns in nitrogen and phosphorus in the Klamath River, USA. Journal of Hydrology 519: 164–176. http://dx.doi.org/10.1016/j.jhydrol.2014.06.025

Otten, T. G., J. R. Crosswell, S. Mackey, and T. W. Dreher. 2015. Application of molecular tools for microbial source tracking and public health risk assessment of a Microcystis bloom traversing 300 km of the Klamath River. Harmful Algae 46: 71–81. http://dx.doi.org/10.1016/j.hal.2015.05.007.

Otten, T. 2017. Application of genetic tools for improved cyanobacterial bloom monitoring in the Klamath River system: Implications for public health monitoring. Prepared by Bend Genetics, LLC., Sacramento, California.

PacifiCorp. 2004a. Analysis of potential Klamath Hydroelectric Project effects on water quality aesthetics for the Klamath Hydroelectric Project (FERC Project No. 2082). Final Technical Report. Prepared by PacifiCorp, Portland, Oregon.

PacifiCorp. 2004b. Water resources for the Klamath Hydroelectric Project (FERC Project No. 2082). Final Technical Report. Prepared by PacifiCorp, Portland, Oregon.

PacifiCorp. 2004c. Screening level determination of chemical contaminants in fish tissue in selected Project reservoirs for the Klamath Hydroelectric Project (FERC Project No. 2082). Final Technical Report. Prepared by PacifiCorp, Portland, Oregon.

PacifiCorp. 2005. Klamath River Water Quality Model Implementation, Calibration, and Validation. Response to November 10, 2005, FERC AIR GN-2. Prepared by PacifiCorp, Portland, Oregon.

PacifiCorp. 2006. Causes and effects of nutrient conditions in the upper Klamath River for the Klamath Hydroelectric Project (FERC Project No. 2082). PacifiCorp, Portland, Oregon.

PacifiCorp. 2008a. 2008 Iron Gate turbine venting study of dissolved oxygen improvement. PacifiCorp, Portland, Oregon.

PacifiCorp. 2008b. Final dataSonde data from below Iron Gate Dam. Water quality preliminary raw data collected with a YSI 6600 DataSonde on the Klamath River downstream from Iron Gate Dam (RM 193.1) from June 20, 2008 through December 23, 2008. Collected by PacifiCorp, Portland, Oregon. http://www.pacificorp.com/es/hydro/hl/kr.html# [Accessed on 3 July 2017].

PacifiCorp. 2008c. Resubmittal of section 401 water quality certification for PacifiCorp Energy's Klamath hydroelectric project (FERC No. 2082), Siskiyou County. Prepared by PacifiCorp, Portland, Oregon for State Water Resources Control Board, Sacramento, California.

PacifiCorp. 2009. Final dataSonde data from below Iron Gate Dam. Water quality preliminary raw data collected with a YSI 6600 DataSonde on the Klamath River downstream from Iron Gate Dam (RM 193.1) from January 16, 2009 through December 31, 2009. Collected by PacifiCorp, Portland, Oregon. http://www.pacificorp.com/es/hydro/hl/kr.html# [Accessed on 3 July 2017].

PacifiCorp. 2010a. Analysis of Microcystin in Fish in Copco and Iron Gate Reservoirs in 2009. Technical Memorandum. PacifiCorp, Portland, Oregon.

PacifiCorp. 2010b. Final dataSonde data from below Iron Gate Dam. Water quality preliminary raw data collected with a YSI 6600 DataSonde on the Klamath River downstream from Iron Gate Dam (RM 193.1) from January 1 thru November 19, 2010. Collected by PacifiCorp, Portland, Oregon. http://www.pacificorp.com/es/hydro/hl/kr.html# [Accessed on 3 July 2017].

PacifiCorp. 2011. Final dataSonde data from below Iron Gate Dam. Water quality preliminary raw data collected with a YSI 6600 DataSonde on the Klamath River downstream from Iron Gate Dam (RM 193.1) from January 1 thru December 31, 2011. Collected by PacifiCorp, Portland, Oregon. http://www.pacificorp.com/es/hydro/hl/kr.html# [Accessed on 3 July 2017].

PacifiCorp. 2012. Final dataSonde data from below Iron Gate Dam. Water quality preliminary raw data collected with a YSI 6600 DataSonde on the Klamath River downstream from Iron Gate Dam (RM 193.1) from January 1 thru December 31, 2012. Collected by PacifiCorp, Portland, Oregon. http://www.pacificorp.com/es/hydro/hl/kr.html# [Accessed on 11 December 2018].

PacifiCorp. 2013. Final dataSonde data from below Iron Gate Dam. Water quality preliminary raw data collected with a YSI 6600 DataSonde on the Klamath River downstream from Iron Gate Dam (RM 193.1) from January 1 thru December 31, 2013. Collected by PacifiCorp, Portland, Oregon. http://www.pacificorp.com/es/hydro/hl/kr.html# [Accessed on 11 December 2018].

PacifiCorp. 2014. Final dataSonde data from below Iron Gate Dam. Water quality preliminary raw data collected with a YSI 6600 DataSonde on the Klamath River downstream from Iron Gate Dam (RM 193.1) from January 1 thru December 31, 2014. Collected by PacifiCorp, Portland, Oregon.
http://www.pacificorp.com/es/hydro/hl/kr.html# [Accessed on 11 December 2018].

PacifiCorp. 2015. Final dataSonde data from below Iron Gate Dam. Water quality preliminary raw data collected with a YSI 6600 DataSonde on the Klamath River downstream from Iron Gate Dam (RM 193.1) from January 1 thru December 31, 2015. Collected by PacifiCorp, Portland, Oregon.
http://www.pacificorp.com/es/hydro/hl/kr.html# [Accessed on 11 December 2018].

PacifiCorp. 2016. Final dataSonde data from below Iron Gate Dam. Water quality preliminary raw data collected with a YSI 6600 DataSonde on the Klamath River downstream from Iron Gate Dam (RM 193.1) from January 1 thru December 31, 2016. Collected by PacifiCorp, Portland, Oregon.
http://www.pacificorp.com/es/hydro/hl/kr.html# [Accessed on 11 December 2018].

PacifiCorp. 2017. Final dataSonde data from below Iron Gate Dam. Water quality preliminary raw data collected with a YSI 6600 DataSonde on the Klamath River downstream from Iron Gate Dam (RM 193.1) from January 1 thru December 31, 2017. Collected by PacifiCorp, Portland, Oregon.
http://www.pacificorp.com/es/hydro/hl/kr.html# [Accessed on 11 December 2018].

Perry, R. W., J. C. Risley, S. J. Brewer, E. C. Jones, and D. W. Rondorf. 2011. Simulating water temperature of the Klamath River under dam removal and climate change scenarios. U.S. Geological Survey Open File Report 2011-1243. U.S. Department of Interior, U.S. Geological Survey, Reston, Virginia.

Raymond, R. 2008. Water quality conditions during 2007 in the vicinity of the Klamath Hydroelectric Project. Prepared by E&S Environmental Chemistry, Inc., Corvallis, Oregon for PacifiCorp Energy, Portland, Oregon.

Raymond, R. 2009. Water quality conditions during 2008 in the vicinity of the Klamath Hydroelectric Project. Prepared by E&S Environmental Chemistry, Inc., Corvallis, Oregon for CH2MHill, Portland, Oregon and PacifiCorp Energy, Portland, Oregon.

Raymond, R. 2010. Water quality conditions during 2009 in the vicinity of the Klamath Hydroelectric Project. Prepared by E&S Environmental Chemistry, Inc., Corvallis, Oregon for PacifiCorp Energy, Portland, Oregon.

Risley, J. C., S. J. Brewer, and R. W. Perry. 2012. Simulated effects of dam removal on water temperatures along the Klamath River, Oregon and California, using 2010 Biological Opinion flow requirements. U.S. Geological Survey Open-File Report 2011-1311.

Scheiff, T., and P. Zedonis. 2011. The influence of Lewiston Dam releases on water temperatures of the Trinity and Klamath rivers, California. April to October 2010. Arcata Fisheries Data Series Report Number DS 2011-22. U. S. Fish and Wildlife Service, Arcata Fish and Wildlife Office, Arcata, California.

Shannon & Wilson, Inc. 2006. Sediment sampling, geotechnical testing and data review report: segment of Klamath River, Oregon and California. Prepared by Shannon & Wilson, Inc., Seattle, Washington for California Coastal Conservancy, Oakland, California.

Sinnott, S. 2008. 2007 Klamath River nutrient summary report. Final Report. Prepared by Yurok Tribe Environmental Program, Water Division, Klamath, California.

Sinnott, S. 2009a. 2006 Nutrient summary report. Final Report. Prepared by Yurok Tribe Environmental Program, Water Division, Klamath, California.

Sinnott, S. 2009b. 2008 Klamath River nutrient summary report. Final Report. Prepared by Yurok Tribe Environmental Program, Water Division, Klamath, California.

Sinnott, S. 2010a. 2009 Klamath River dataSonde report. Final Report. Prepared by Yurok Tribe Environmental Program, Klamath, California.

Sinnott, S. 2010b. 2009 Klamath River nutrient summary report. Final Report. Prepared by Yurok Tribe Environmental Program, Klamath, California.

Sinnott, S. 2011a. 2010 Klamath River continuous water quality monitoring summary report. Final Report. Prepared by Yurok Tribe Environmental Program, Water Division, Klamath, California.

Sinnott, S. 2011b. 2010 Klamath River nutrient summary report. Final Report. Prepared by Yurok Tribe Environmental Program, Water Division, Klamath, California.

Sinnott, S. 2012a. 2011 Klamath River continuous water quality monitoring summary report. Final Report. Prepared by Yurok Tribe Environmental Program, Water Division, Klamath, California.

Sinnott, S. 2012b. 2011 Klamath River nutrient summary report. Final Report. Prepared by Yurok Tribe Environmental Program, Water Division, Klamath, California.

State Water Board (State Water Resources Control Board). 2010. 2010 California 303(d) list of water quality limited segments, Category 5. Final 2010 Integrated Report (CWA Section 303(d) List/ 305(b) Report). State Water Resources Control Board, Sacramento, California.
http://www.waterboards.ca.gov/water_issues/programs/tmdl/2010state_ir_reports/category5_report.shtml (accessed 4 July 2017).

State Water Board. 2017. Final Staff Report, Including Substitute Environmental Documentation for Part 2 of the Water Quality Control Plan for Inland Surface Waters, Enclosed Bays, and Estuaries of California – Tribal and Subsistence Fishing Beneficial Uses and Mercury Provisions. State Water Resources Control Board, Sacramento, California.
http://www.waterboards.ca.gov/water_issues/programs/mercury/docs/hg_SR_final.pdf (accessed 3 July 2017).

State Water Board, California Department of Public Health (CDPH) and Office of Environmental Health and Hazard Assessment (OEHHA). 2010. Cyanobacteria in California recreational water bodies: providing voluntary guidance about harmful algal blooms, their monitoring, and public notification. Blue Green Algae Work Group of the State Water Resources Control Board, the California Department of Public Health, and Office of Environmental Health and Hazard Assessment.

Stein, R.A., P. E. Reimers, and J. D. Hall. 1972. Social interaction between juvenile coho (*Oncorhynchus kisutch*) and fall Chinook salmon (*Oncorhynchus tshawytscha*) in Sixes River, Oregon. Journal of the Fisheries Research Board of Canada 29: 1,737–1,748.

Stillwater Sciences. 2009. Dam removal and Klamath River water quality: a synthesis of the current conceptual understanding and an assessment of data gaps. Technical Report. Prepared by Stillwater Sciences, Berkeley, California for State Coastal Conservancy, Oakland, California.

Stillwater Sciences. 2010. Anticipated sediment release from Klamath River dam removal within the context of basin sediment delivery. Final Report. Prepared by Stillwater Sciences, Berkeley, California for State Coastal Conservancy, Oakland, California.

Sturdevant, D. 2010. Water quality standards review and recommendations: arsenic, iron, and manganese. Draft Report. Prepared by Oregon DEQ, Water Quality Standards Program, Portland, Oregon.

Sullivan, A. B., M. L. Deas, J. Asbill, J. D. Kirshtein, K. Butler, M. A. Stewart, R. E. Wellman, and J. Vaughn. 2008. Klamath River water quality and acoustic doppler current profiler data from Link River Dam to Keno Dam, 2007. Open-File Report 2008-1185. Prepared by U.S. Department of Interior, U.S. Geological Survey, Reston, Virginia in cooperation with the Bureau of Reclamation.

Sullivan, A. B., M. L. Deas, J. Asbill, J. D. Kirshtein, K. Butler, and J. Vaughn. 2009. Klamath River water quality data from Link River Dam to Keno Dam, Oregon, 2008. U.S. Geological Survey Open File Report 2009-1105. Prepared by the U.S. Department of the Interior, U.S. Geological Survey, Reston, Virginia.

Sullivan, A. B., S. A. Rounds, M. L. Deas, J. R. Asbill, R. E. Wellman, M. A. Stewart, M. W. Johnston, and I. E. Sogutlugil. 2011. Modeling hydrodynamics, water temperature, and water quality in the Klamath River upstream of Keno Dam, Oregon, 2006–2009. Scientific Investigations Report 2011-5105. Prepared by Oregon Water Science Center, U.S. Geological Survey, Portland, Oregon in cooperation with the Bureau of Reclamation.

Sullivan, A. B. and S. A. Rounds. 2016. Modeling water quality, temperature, and flow in Link River, South-Central Oregon. U.S. Geological Survey Open-File Report 2106-1146. http://dx.doi.org/10.3133/ofr20161146.

USBR (United States Bureau of Reclamation). 2010. Quality assurance project plan sediment contaminant study, Klamath River sediment sampling program, JC Boyle,

Copco-1, Copco-2, and Iron Gate Reservoirs; Klamath River Estuary. Prepared for Mid-Pacific Region, U. S. Bureau of Reclamation, Technical Service Center, CO.

USBR. 2011. Sediment Chemistry Investigation: Sampling, Analysis, and Quality Assurance Findings for Klamath River Reservoirs and Estuary, October 2009 – January 2010. In Support of the Secretarial Determination on Klamath River Dam Removal and Basin Restoration, Klamath River, Oregon and California. Prepared for Mid-Pacific Region, U. S. Bureau of Reclamation, Technical Service Center, CO.

USBR. 2012. Hydrology, Hydraulics and Sediment Transport Studies for the Secretary's Determination on Klamath River Dam Removal and Basin Restoration. Technical Report No. SRH-2011-02. Prepared for Mid-Pacific Region, U. S. Bureau of Reclamation, Technical Service Center, CO.

USEPA (U. S. Environmental Protection Agency). 1978. National eutrophication survey. Report on Iron Gate Reservoir, Siskiyou County, California. EPA Region 9. Working Paper No. 749.

USEPA. 2000. National guidance: guidance for assessing chemical contaminant data for use in fish advisories. Volume 2: risk assessment and fish consumption limits. Third edition. EPA 823-B-00-008. Office of Science and Technology, Office of Water, USEPA, Washington, D.C.

USEPA. 2003. Ambient aquatic life water quality criteria for tributyltin (TBT) – Final. EPA 822-R-03-031. Health and Ecological Criteria Division, Office of Science and Technology, Office of Water, USEPA, Washington, D.C.

USEPA. 2010. Subject: compilation and discussion of sediment quality values for dioxin, and their relevance to potential removal of dams on the Klamath River. Memorandum from B. Ross, Region 9 Dredging and Sediment Management Team and E. Hoffman, Region 10 Environmental Review and Sediment Management Unit, USEPA, San Francisco, California to D. Lynch, USGS, and R. Graham, USBR.

USEPA. 2013. Aquatic life ambient water quality criteria for ammonia – freshwater 2013. EPA 822-R-13-001. Office of Science and Technology, Office of Water, USEPA, Washington, D.C.

USDA Forest Service. 2004. Cumulative watershed effects analysis for the Klamath National Forest. Quantitative models for surface erosion, mass wasting, and ERA/TOC.

USFWS (U. S. Fish and Wildlife Service). 2008. Corrected Sonde database. http://www.fws.gov/arcata/fisheries/activities/waterQuality/klamathWQ_reports.html [Accessed on July 2008].

USGS (U. S. Geological Survey). 2011. USGS water data for the nation. U.S. Department of the Interior, U.S. Geological Survey. http://waterdata.usgs.gov/nwis [Accessed on January 2011].

USGS NGS (U.S. Geological Survey National Geochemical Survey). 2008. The National Geochemical Survey – Database and Documentation. U.S. Geological Survey

Open-File Report 2004-1001 Version 5.0. Available at:
https://mrdata.usgs.gov/geochem/map-us.html#home

Wallace, M. 1998. Seasonal water quality monitoring in the Klamath River estuary,
1991–1994. Administrative Report No. 98-9. California Department of Fish and Game,
Inland Fisheries, Arcata, California.

Ward, G.H. and N. E. Armstrong. 2006. Subject: hydrological features of Lower
Klamath River and tributaries. Memorandum to U.S. Fish & Wildlife Service, Arcata
Office from University of Texas at Austin. 15 January (revised 25 February and 1
March).

Ward, G. and N. Armstrong. 2010. Assessment of primary production and associated
kinetic parameters in the Klamath River. Draft Report. Prepared for the USFWS, Arcata
Fish and Wildlife Office, Arcata, California.

Watercourse Engineering, Inc. 2011a. Klamath River baseline water quality sampling,
2009 Annual Report. Prepared for the KHSA Water Quality Monitoring Group.

Watercourse Engineering, Inc. 2011b. Klamath River baseline water quality sampling,
2010 Annual Report. Prepared for the KHSA Water Quality Monitoring Group.

Watercourse Engineering, Inc. 2012. Klamath River baseline water quality sampling,
2011 Annual Report. Prepared for the KHSA Water Quality Monitoring Group.

Watercourse Engineering, Inc. 2013. Klamath River baseline water quality sampling,
2012 Annual Report. Prepared for the KHSA Water Quality Monitoring Group.

Watercourse Engineering, Inc. 2014. Klamath River baseline water quality sampling,
2013 Annual Report. Prepared for the KHSA Water Quality Monitoring Group.

Watercourse Engineering, Inc. 2015. Klamath River baseline water quality sampling,
2014 Annual Report. Prepared for the KHSA Water Quality Monitoring Group.

Watercourse Engineering, Inc. 2016. Klamath River baseline water quality sampling,
2015 Annual Report. Prepared for the KHSA Water Quality Monitoring Group.

WHO (World Health Organization). 1999. Concise International Chemical Assessment
Document 14 tributyltin oxide. World Health Organization, Geneva, Switzerland.

YTEP (Yurok Tribe Environmental Program). 2004. Water year 2002 (WY02) report, 1
October 2001–30 September 2002. Final Report. Prepared by Yurok Tribe
Environmental Program, Klamath, California.

YTEP. 2005. Water year 2004 (WY04) report, 1 October 2003–30 September 2004.
Final Report. Prepared by Yurok Tribe Environmental Program, Klamath, California.

YTEP. 2014. Aug 5-6, 2014 Microcystin Results. Yurok Tribe Environmental Program
(YTEP) Memorandum. Prepared by Yurok Tribe Environmental Program, Klamath,
California.

YTEP. 2015. July 28 & 29 BGA Results. Yurok Tribe Environmental Program (YTEP) Memorandum. Prepared by Yurok Tribe Environmental Program, Klamath, California.

Zedonis, P. and R. Turner. 2010. An investigation of water quality above, within, and below Iron Gate Reservoir during the fall of 2004. 2005 Draft Report. U.S. Fish and Wildlife Service, Arcata Fish and Wildlife Office, Arcata Fisheries Technical Report., Arcata, California.

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